# FINAL REPORT

Reptile, Amphibian, Breeding Bird, and Invertebrate Response to Prescribed Fires and Mechanical Fuel reduction Treatments in an Upland Hardwood Forest

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C: Control	

## Keywords

MB: Mechanical + burn

Amphibians, beetles, breeding birds, habitat, fire severity, fuel reduction, invertebrates, pollinators, prescribed burn, reptiles

M: Mechanical SE: Standard error

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#### **Abstract**

Repeated fire and fire surrogate treatments for fuel reduction, forest restoration, and wildlife conservation affect forest structure, in turn influencing vertebrate and invertebrate taxa response, but long-term investigation of these relationships is lacking. We experimentally assessed long-term (2001-2016) effects of repeated mechanical and prescribed fire treatments on herpetofauna, breeding bird, and invertebrate communities in upland hardwood forest. Treatments were untreated controls (C); mechanical understory removal (twice) (M); dormant season burns (four times) (B), and; mechanical understory removal followed a year later by a high-severity burn and three subsequent burns (MB). Initial burns were hotter in MB than B, resulting in heavy tree mortality, increased canopy openness, greater shrub density, and abundant snags lasting several years. Subsequent, lower-intensity prescribed burns in MB maintained the open-forest structure. In contrast, four repeated low-intensity burns (B) resulted in delayed mortality of smaller trees, or some overstory trees in 'hot spots,' leading to increased structural heterogeneity over time. Despite reduction in shrub stem density after each repeated fuel reduction treatment, shrubs recovered rapidly, and eventually exceeded pretreatment levels in MB, as top-killed trees and shrubs resprouted. Leaf litter depth decreased after burns in B and MB, but recovered rapidly as leaves dropped from deciduous trees each fall. Post-treatment understory reductions in M were relatively small and transient. Restoration to an open woodland was not achieved, likely due to heavy canopy retention in M and B, and resprouting of top-killed trees and shrubs in MB that inhibited establishment of grasses and herbaceous plants. No common herpetofaunal species, including terrestrial salamanders were adversely affected by any fuel reduction treatment. Five-lined skink (Plestiodon fasciatus) and eastern fence lizard (Sceloporus undulatus) captures were greater in MB, where open conditions likely provided more opportunity for thermoregulation and successful reproduction. A trend of increased fivelined skink and eastern fence lizard captures over time in MB suggested that response may be delayed, or repeated burning enhances habitat conditions for these species over time. Increased capture rates of juvenile eastern fence lizards after a third and fourth prescribed fire in MB indicates that repeated burns may further increase habitat suitability for successful recruitment. Bird density and species richness increased in MB within three breeding seasons of initial highseverity burns, and remained greater throughout subsequent burns. Bird richness was greater in B than C or M but did not differ from MB, likely due to more canopy gaps in B. Bird richness in MB increased due to an influx of species associated with young forest conditions, with little change in generalist-, or most mature forest species. Density increases in MB were most

pronounced in the shrub- and cavity-nesting bird guilds; ground-nester density was lower in MB than C or M. Repeated burns in MB impeded forest maturation and maintained young forest conditions, but did not provide additive effects to breeding bird communities. Mechanical understory removal (M) had little effect on bird density and richness. Bee abundance (driven by Lasing lossum spp.), diversity, and genera richness on the forest floor was greater in one or both burn treatments compared to M or C. Lasioglossum and Bombus bees were more abundant in MB than C. The abundance of common wasp families did not differ among treatments. Syrphid fly abundance on the forest floor was greater in M, B, and MB compared to C. Treatment differences were not detected for flower-visiting taxa trapped in the midstory, but bees were more abundant in the midstory than the forest floor. Among wasps, Pompilidae was more abundant on the forest floor; Vespula spp. and Dolichovespula maculata L. were more abundant within the midstory. Hesperiidae butterflies were more abundant at the forest floor. Abundance of several Coleoptera families differed among treatments, but responses differed. For example, Nitidulidae was more abundant in C than B or MB, whereas Mordellidae was more abundant in B than C or M. No common Coleoptera species differed in abundance among treatments. Fungivores and phytophagous/predator beetles were more abundant in C than MB; phytophagous and coprophagous beetles were more abundant in B than in M. Species diversity within common families (Carabidae, Cerambycidae, Cleridae, and Scarabaeidae) did not differ among the treatments. Coleoptera communities as a whole remained abundant and diverse after repeated fuel reduction treatments. Our study illustrates the importance of long-term studies that can address potentially delayed responses to forest disturbances, and potential additive effects of repeated disturbances. Different responses among species emphasize the importance of including multiple taxa when assessing effects of forest disturbances on wildlife, and give perspective on how the definition of forest health may vary depending on target taxa.

## **Objectives**

The Appalachian site of the Fire and Fire Surrogate Study (FFS) has been maintained since 2001 in partnership with the North Carolina Wildlife Resources Commission (NCWRC), with repeated fuel reduction treatments, and periodic measurement of vegetation, fuels, and soils. Our earlier studies indicated that high-severity fire substantially altered forest structure and wildlife communities, whereas one or two low-intensity burns or a mechanical treatment had fewer and transient effects. However, those short-term studies could not address longer-term impacts, or whether effects might be cumulative with repeated fuel reduction treatments. We proposed to take the next step by measuring the longer-term impacts of repeated (total four burns and two mechanical understory reductions) fuel reduction treatments on herpetofauna, breeding birds, and invertebrates in relation to key habitat parameters, and develop guidelines to assess restoration activities targeted at these faunal communities. We hypothesized:

- 1) Repeated burning will maintain open, young forest conditions and associated microclimatic conditions in MB, and delayed or new tree mortality in B will create a heterogeneous canopy structure with patches of open habitat;
- 2) Salamander abundance will be lowest in MB due to reduced leaf litter and less shade that affects microclimate, but also lower in B compared to unburned controls (C);
- 3) Lizard abundance will be greatest in MB but also higher in B compared to C;

- 4) Breeding bird abundance and species richness will remain higher in MB than other treatments and continue to increase in B compared to C, with co-occurrence of species associated with young- and mature forest conditions;
- 5) Insect pollinator abundance and species richness will be highest in MB followed by B compared to other treatments due to greater availability of potential nesting structure for bees and more food resources;
- **6**) Abundance of some ground-dwelling taxa will differ among treatments due to differences in microhabitat structure such as leaf litter cover and bare ground.

Our questions and objectives related to the JFSP-identified task statement by assessing the effectiveness of repeated fuel reduction treatments on ecosystem restoration, as measured by changes to forest structure and reptile, amphibian, breeding bird, and invertebrate communities.

## **Background**

Prescribed burning is an important land management tool for upland hardwood forests, with fuel reduction, ecosystem restoration, and wildlife habitat improvement often cited as primary goals. Mechanical fuel reduction by cutting shrubs and small trees (also termed "fire surrogates") is sometimes used instead of prescribed burns to reduce risks to property, safety, and air quality associated with fire (Waldrop et al. 2016). Prescribed burns are usually conducted in winter, and under restrictive fuel and weather conditions that generally result in low-intensity burns to minimize safety risks and potential damage to timber. Accordingly, post-fire changes to forest structure – a primary driver of wildlife community composition (Greenberg et al. 2011; Greenberg et al. 2015a) – are often limited to transitory reductions in shrub and leaf litter cover, with little overstory mortality (Waldrop et al. 2016). Despite substantial investment of time and funding for ecosystem restoration, stated goals are often vague, with little guidance available regarding methods to achieve goals, the temporal scale required, or metrics to assess effectiveness.

Historic fire frequency and landscape occurrence in upland hardwood forest were strongly associated with population centers of Native Americans and later European settlers, who used fire to clear land, hunt, and increase forage for game or livestock (Spetich et al. 2011; Greenberg et al. 2015a); lightning-caused (non-anthropogenic) fires were rare (Greenberg et al. 2015b). Additionally, fire frequency and severity in upland hardwood forests was likely variable, mediated by weather, fuels, and topography, particularly in mountainous terrain such as the southern Appalachians. Accordingly, effects of fire or fire surrogates on vertebrates and invertebrates is not well understood, especially in hardwood forests. Fire management for restoration or wildlife conservation requires an understanding of how different taxa, species, or guilds respond to burning in different forest types, at different frequencies and severities, and over time.

Increasingly, "restoration burns" are conducted across large landscapes of diverse topography and fuel loads with incomplete knowledge of how different frequencies, seasons, or severities of burns affect biotic communities. Short-term studies indicate that low-intensity dormant-season prescribed burns in upland hardwood forest has a minimal and transient effect on forest structure or wildlife (Moorman et al. 2011; Greenberg et al. 2014). However, repeated burning could have additive effects on habitat attributes, such as canopy cover, shrub density, or leaf litter depth, and associated changes in forest floor microclimate, food, and cover resources

that might alter suitability for various wildlife taxa over the longer-term. Additionally, there could be a time lag in fire-induced changes, most notably delayed mortality of overstory trees (Waldrop et al. 2016). Prescribed fire is not a precise forest management tool; weather, fuel types and volumes, vegetation structure, topography, and ground moisture affect fire intensity and consequent post-fire tree mortality (Knapp et al. 2009). Although changes in forest structure associated with high-severity fires may affect some herpetofaunal (Matthews et al. 2010, Fouts et al. 2017), breeding bird (Greenberg et al. 2013), or pollinating insect (Campbell et al. 2007) species in the short-term, longer-term studies are needed to document delayed changes to vegetation structure and associated wildlife communities, in relation to repeated burning and fire severity.

We conducted a series of earlier studies through the National FFS infrastructure (at the same study sites reported here) examining the responses of breeding bird (Greenberg et al. 2013), reptile and amphibian (Matthews et al.; Greenberg et al. 2017), invertebrate (Greenberg et al. 2010, Campbell et al. 2007), and other wildlife communities to prescribed burning, mechanical fuel reduction, or MB fuel reduction treatments after a single treatment, and again after a second burn in the two burn treatments. Our earlier results indicated that wildlife and invertebrate communities and species respond differently to habitat conditions created by prescribed burning, and impacts were greater in the MB treatment, where initial burns were high-severity. Additionally our studies suggested that salamanders may show a delayed negative response to high-severity burning (Matthews et al. 2010), and additional breeding bird species may be attracted to sites that are repeatedly burned at a low intensity, as delayed or cumulative tree mortality creates more snags and canopy gaps. Burned sites may also positively influence some insect pollinators by providing ground-nesting habitat (ants and bees) and a greater diversity or abundance of flowering herbaceous plants (Campbell et al. 2007).

Here, we use the same experimental design and study sites to assess longer-term response of herpetofaunal, breeding bird, insect pollinators, and ground-dwelling invertebrate communities to a third and fourth prescribed fire in the two burn treatments, and a second mechanical felling of the understory (2012) in the M treatment. This multi-phased study of different initial fire severities provides a unique temporal and spatial perspective on the use of prescribed fire for ecosystem restoration. Additionally, it provides an opportunity to assess ecosystem restoration and fuel reduction efforts at a longer temporal scale, and develop metrics to gauge effectiveness based on species composition of herpetofaunal, breeding bird, insect pollinator, and ground-dwelling arthropod communities in relation to time and habitat conditions.

#### Methods

## **Study Area**

We conducted our study on the 5,841-ha Green River Game Land (35° 17'0900N, 82° 19'42"W, blocks 1 and 2; 35°15'42"N, 82° 17'27"W, block 3) in Polk County, North Carolina, USA (Fig. 1). The Game Land is in the mountainous Blue Ridge Physiographic Province of western North Carolina, characterized by a temperate climate with warm, humid summers and mild winters. Average annual precipitation is 1,638 mm and is distributed evenly throughout the year, and average annual temperature is 17.6° C. Soils are primarily sandy loam (NCWRC 2014). Elevation ranged from approximately 366–793 m. The Game Land was 97% forested, and had been managed for wildlife conservation since its purchase in the 1950s (NCWRC 2014). The upland hardwood forest was composed mainly of oaks (*Quercus* spp.) and hickories (*Carya* 

spp.). Shortleaf pine (*Pinus echinata*) and Virginia pine (*P. virginiana*) were on ridgetops, and white pine (*P. strobus*) occurred in moist coves. Forest age within experimental units ranged from about 85 to 125 years. Predominant shrubs were mountain laurel (*Kalmia latifolia*) along ridge tops and on upper southwest-facing slopes, and rhododendron (*Rhododendron maximum*) in mesic areas. Prior to the first prescribed burns in 2003, none of the sites had been thinned or burned.

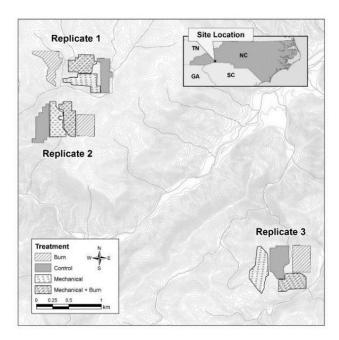
### **Sampling Design**

We selected three study areas (blocks) within the Game Land (Fig. 1) based on size (capacity to accommodate four experimental units each), forest age, cover type, and management history, to ensure consistency in baseline conditions among the treatments. Perennial streams bordered and (or) traversed all three replicate blocks. Minimum size of experimental units within blocks was 14 ha, to accommodate 10-ha core areas with 20-m wide buffers around each. Dirt roads or fire lines separated some of the experimental units but did not traverse any, and wooded trails traversed some experimental units.

We assigned three fuel reduction treatments and an untreated control randomly within each of the three study blocks, for 12 experimental units. Treatments were: 1) repeated prescribed burns (B; four times in February or March 2003, 2006, 2012, and 2015); 2) repeated mechanical felling of all shrubs and small trees >1.4 m tall and <10.2 cm in diameter at breast height (dbh) with a chainsaw (M; twice in winters 2001-2002 and 2011-2012); and 3) initial mechanical cutting of the understory (winter 2001-2002) followed by four prescribed burns (MB; burns timed with B units; Table 1). Cut fuels were left scattered onsite resulting in little or no vertical structure initially, with subsequent recovery in M.

During the first prescribed burns (2003), flame lengths of 1--2 m occurred throughout all burn units, but flame lengths reached up to 5 m in localized spots within blocks, where topography or intersecting flame fronts contributed to erratic fire behavior (Waldrop et al. 2010). Loading of fine woody fuels on MB units, where the shrub layer was felled, was approximately double that on C and B sites. Average fire temperature at 30 cm aboveground was much hotter in MB than B (370°C and 180°C, respectively). The second burn (2006) was less intense, with flame lengths generally <1.5 m. Average temperature 30 cm aboveground was 155°C in B units and 222°C in MB units (Waldrop et al. 2016). We did not measure fire temperatures in the third and fourth burns, but we observed that they were low-intensity with flame lengths <2 m.

Fig. 1. Study area map.



**Table 1.** Timing of repeated, dormant-season fuel reduction treatments applied to experimental units (n = 3 per treatment), and years (growing season) when we sampled herpetofauna, breeding birds, invertebrates, and forest structure within the Green River Game Land, Polk County, North Carolina, USA, 2001–2016. Treatments were a mechanical understory removal followed by a high-severity burn (MB), and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (M; two applications); and controls (C; n = 3 per treatment).

Treatment	Pretreatment 2001	Winter '01-'02	Winter '02-'03	Spring-Summer	Winter '05-'06	Spring- Summer	Winter '11-'12	Spring- Summer	Winter '14-'15	Spring- Summer
Control (C)										
Mechanical (M)		M					M			
Burn (B)			В		В		В		В	
Mechanical+burn (MB)		M <sup>1</sup>	$\mathbf{B}^2$		$\mathbf{B}^2$		$\mathbf{B}^2$		$\mathbf{B}^2$	
Sample Periods										
Herpetofaunal Trapping <sup>3</sup>				2003, 2004		2006, 2007		2014		2015, 2016
Breeding bird sampling	2001			2003, 2004, 2005		2006, 2007, 2009, 2011		2012, 2014		2015, 2016
Invertebrate sampling				2003, 2004 <sup>4</sup>				2014		2015, 2016
Vegetation sampling	2001			2002 & 2004 (M) 2003 & 2005 (B,MB,C)		2006 (all trts) 2011 (all trts)		2012 (all trts) 2014 (all trts)		2015 (all trts) 2016 (all trts)

<sup>&</sup>lt;sup>1</sup> Mechanical understory removal only

<sup>&</sup>lt;sup>2</sup> Prescribed burn only

<sup>&</sup>lt;sup>3</sup> Traps opened continuously and concurrently as follows: 5 May-2 July and 28 July-18 August 2003; 7 May-16 August 2004; 17 May-16 August 2006; 15 May-13 August 2007; 21 May-11 August 2014; 14 May-9 August 2015; 16 May-5 August 2016.

<sup>&</sup>lt;sup>4</sup> This sampling period not included in this report; see Campbell et al. 2007

#### **Field Measurements**

*Habitat/Vegetation measurements*: We measured live tree and snag (≥10 cm dbh) density, live tree (BA), shrub (woody stems >1.4 m ht and <10 cm dbh) stem density, and leaf litter depth during most years when breeding birds, herpetofauna, or pollinating insects were sampled: pretreatment (2001; all treatments); one growing season after all initial treatments were implemented (2002 for M, and 2003 for B, MB, and C treatment units); three growing seasons after initial treatments (2004 for M; 2005 for B, MB, and C); five growing seasons after the mechanical understory removal in M and one growing season after a second burn in B and MB (2006; all treatments); 10 growing seasons after the mechanical understory removal in M and six growing seasons after the second burn in B and MB (2011; all treatments); one growing season after a second mechanical understory removal in M and a third burn in B and MB (2012; all treatments); two growing seasons after a second mechanical understory removal in M and three growing seasons after a third burn in B and MB (2014; all treatments); and, three and four growing seasons after the second mechanical understory removal in M, and one and two years after a fourth burn in B and MB (2015 and 2016; all treatments) (Table 1). Tree and snag density were measured within 10, 0.05-ha (10 x 50 m) plots located at 50 x 50 m intervals starting from a randomly selected grid-point origin within each experimental unit (Waldrop et al. 2016). Shrub stem density (including all stems within sprout clumps) was measured within 20, 1m<sup>2</sup> quadrats within each vegetation plot. Leaf litter depth was measured using a meter stick at three locations along each of three randomly oriented, 15-m transects originating at grid points that were spaced at 50-m intervals throughout each experimental unit. Litter depth was measured only in B and MB in 2011 and 2012 (before and after the third burn) and was not measured at all in 2016. We measured percent canopy openness at drift fence-level during years when herpetofauna were trapped (2003, 2004, 2006, 2007, 2014, 2015, 2016), using a spherical densiometer, at the center bucket of each randomly located drift fence array (see below) within each experimental unit during summer (leaf on) as a crude metric of understory light and microclimate. For each habitat feature measured we used the average (plots, quadrats, or transects) across each experimental unit (n = 3 per treatment) in our statistical analyses.

**Reptile and amphibian trapping:** We trapped reptiles and amphibians using drift fence arrays that were open continuously and concurrently during May-August in all units, after all initial treatments were fully implemented (2003 and 2004); after a second prescribed burn in B and MB (2006 and 2007); after a second mechanical thinning in M and a third prescribed burn in B and MB (2014), and; after a fourth prescribed burn in B and MB (2015–2016; Table 1). We also trapped briefly prior to treatments (2001) but excluded those data here because the season (late summer-fall) and length of trapping period (56 nights) were not comparable to subsequent data. However, pre-treatment data indicated that species richness and capture rates of all taxa were similar among treatments (Greenberg and Waldrop 2008). The number of operational drift fence arrays was consistent across all treatment units each year but differed among years. We installed two arrays for trapping in 2003–2004. We started the 2006 trapping season with two arrays but added a third to each unit; the original two were opened concurrently during mid-season 2006, and all three arrays were operational for the 2007 trapping season. We pulled out all arrays after the 2007 season but re-installed them in 2014 at the same or nearby location of original trap arrays. We started the 2014 trapping season with three arrays but added a fourth to each unit; the fourth array was opened in all units a week after trapping began in 2014, and all were operational for the entire 2015 and 2016 trapping seasons. The number of array nights (number of arrays ×

the number of nights operational) totaled 20,772 across all years, and ranged 158–341 per treatment unit each year. We placed drift fence arrays randomly ≥100 m apart within treatment units. Arrays were constructed with three, 7.6-m, 50-cm high sections of aluminum flashing positioned at approximately 120° angles (in a Y configuration), with 1, 19-L bucket buried at the center, and at the end of each arm, for four pitfall traps per array (Fig. 2). We placed a doubleended funnel trap, constructed from aluminum screening, along both sides of each arm for 6 funnel traps per array (Fig. 2). We drilled holes in the bottoms of pitfalls to prevent flooding. We shaded all traps with a small board, and placed a sponge in pitfall traps moistened as needed to provide cover and humidity for captured animals; frequently flooded buckets also contained a small piece of sponge or styrofoam for flotation. We checked all drift-fence arrays every 1–3 days and every day following a rain event. We identified, measured, and marked individuals by year and treatment by toe-clips (lizards, frogs, and salamanders), scale-clips (snakes), or scutenotching (turtles) except in 2006–2007, when we instead marked amphibians using Visible Implant Elastomer. We standardized all herpetofaunal capture data for differences in trapping effort among years by using captures per 100 array nights (one array night included four pitfall and six funnel traps open for one night).

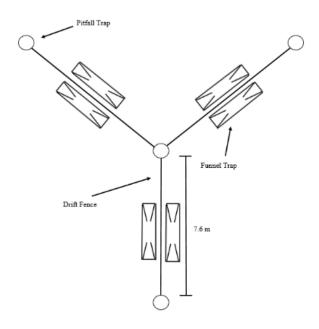
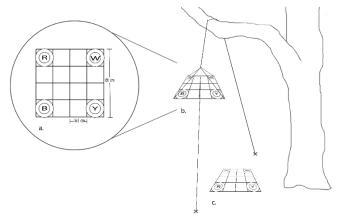


Fig. 2. Herpetofaunal trap array.

*Breeding bird sampling*: We surveyed breeding bird communities using three, 50-m radius (0.785-ha area) point counts spaced 200 m apart in each experimental unit (Ralph et al. 1993). Each point was surveyed for 10 minutes during three separate visits between 15 May and 30 June during each year sampled. Bird surveys were conducted in 2001 (pretreatment); 2003, 2004, and 2005 (all initial treatments were implemented prior to breeding season 2003); in 2006, 2007, 2009, 2011 (after a second burn in B and MB, March 2006); in 2012 and 2014 (after a second mechanical understory removal in M, January-February 2012, and a third burn in B and MB,

**Fig. 3**. Schematic diagram of colored pan trap set-up within the midstory and on the forest floor for invertebrate trapping.



March 2012); and in 2015 and 2016 (after a fourth burn in B and MB, March 2014) (Table 1). Point counts were conducted within four hours of sunrise. All birds seen or heard within a 50-m radius were recorded. Point count times were rotated among the three visits to each experimental unit to avoid time-of-day bias. Each unit was surveyed early-, mid-, and late-season within the 6-week survey period to avoid bias associated differences in singing rates as breeding season progressed. Most point counts were conducted by a single observer (J. Tomcho; total three observers during the entire study period). We did not estimate detectability of different bird species (Alldredge et al. 2008), and assumed that bird detection error was minimal and consistent among units due to a small (50 m) point count radius, one primary observer, multiple survey points, repeated surveys within each unit, and timing of surveys across time of day and breeding season. Relative density of birds within experimental units was calculated by averaging across the three surveys and three point counts (nine observation periods per unit) for each year, and extrapolating the average number per point count to number per 10 ha. Species richness represented the total number of species detected during all three visits and point counts in each experimental unit each year.

Invertebrate sampling: We used sets of colored pan traps filled with soapy water to sample pollinating insects (Campbell and Hanula 2007, Campbell et al. in press). A bowl set consisted of a red, blue, white, and yellow bowl placed at each corner of a 66 cm square of metal remesh (Nucoar) (Fig. 3). At each of two locations  $\geq 50$  m apart within each treatment unit we placed a bowl set (one square wire remesh with four bowls) on the forest floor, and hoisted another set ( $\leq 10$  m horizontal distance away from bowl set) into the midstory at a mean height of  $9.1\pm0.3$  m (Fig. 3). We also established two pitfall trap arrays spaced  $\geq 50$  m apart within each treatment unit to sample ground-dwelling invertebrates. Arrays consisted of a 118 ml cup filled half-way with soapy water, with three 30 cm long aluminum flashing drift fences trenched into the ground, radiating from the center of the cup and oriented at  $120^{\circ}$  to the neighboring drift fence. Bowl sets and pitfall traps were deployed for 72 hours at monthly intervals (May/June- September/October) from 2014-2016. We conducted a total of 11 colored pan trapping periods (three in 2014; four in 2015 and 2016) and 12 pitfall trapping periods (four each year) during the three year study.

**Data Analysis:** We used general linear model one-way analysis of variance (ANOVA) (all three years combined) or, if data could not be normalized a Kruskal-Wallis one-way nonparametric

analysis of variance to analyze treatment effects on invertebrates. We used a mixed model repeated-measures analysis of variance (ANOVA) with treatment and year modeled as fixed effects and block as a random effect to analyze treatment effects on measured habitat variables; capture rate and species richness of reptiles and amphibians; species richness and density (total, species (if  $\geq$  50 total observations), and tree-, shrub-, cavity-, or ground-nesting guilds) of breeding birds. Logarithmic and square root data transformations were used when appropriate in these analyses. Treatment, year, or treatment x year interaction differences were considered significant with an overall experimental  $\alpha$  of < 0.05.

#### **Results**

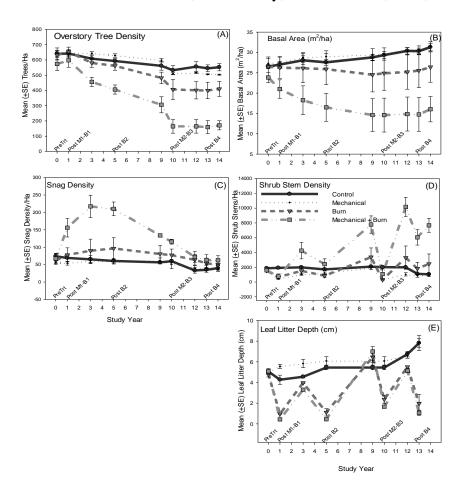
### Habitat/Vegetation

Live tree density was lower in MB than in other treatments, and a significant treatment x year interaction was detected (Table 2; Fig. 4a). Declines in live tree density were evident within two years of the initial high-severity burns in MB, and had declined significantly (29.3% lower than pretreatment) by 2006, and by 71.3% in 2012, after the third burn (Table 2; Figure 3a). In B, live tree density was significantly reduced by 2005 (8.1% lower than pretreatment), and further significant reductions (35.7%) were evident by 2011. Live tree basal area differed among years and was lower in MB than C or M; a treatment x year effect was detected (Table 2; Fig. 4b). In MB, basal area was significantly reduced (30.8% lower than pretreatment) basal area by 2006, and further reductions by 2011. Snag density was greater in MB than C or M and differed among years; an interaction effect was detected (Table 2; Fig. 4c). Snag density more than doubled in MB beginning after the high-severity burn (2003), peaked within 3-4 growing seasons (222.8% more snags in 2005 than pretreatment), and subsequently decreased as snags fell; by 2014 there were no significant (p<0.008) differences in snag densities among treatments. In B, snag availability differed only marginally from other treatments; snag density increased 55.9% from pretreatment levels by 2006 (one growing season after the second burn), and subsequently decreased. In C, snag availability decreased gradually from an average of 74/ha in 2001, to 40/ha in 2016. Shrub stem density differed among years and was greater in MB than M; a treatment x year interaction was detected (Table 2; Fig. 4d). Shrub stem density changed over time within all treatments, and treatments differed within all years except 2001 (pretreatment) and 2006 (one growing season after the second burn). In M, shrub stem density was significantly reduced by each mechanical understory removal (2003 and 2012), and rapidly recovered to approximately pretreatment levels. In B, shrub stem density was significantly decreased by each burn, also recovering rapidly to approximately pretreatment levels. In MB, shrub stem density was also significantly reduced by each burn, but only two of the four burns (2003 and 2012) reduced stem densities to below pre-treatment levels. Recovery was rapid, with shrub stem densities far exceeding pretreatment levels within 1–3 growing seasons of a burn (2005, 2011, 2014, 2015, and 2016). For example, shrub stem densities were 537.4% greater than pretreatment three growing seasons after the third burn (2014); immediately after the fourth (2015) burn, stem densities still exceeded pretreatment levels by 279.1% (Figure 3d). Leaf litter depth was lower in MB than other treatments, and lower in B than C or M; a treatment x year interaction effect was detected (Table 2; Fig. 4e). Litter depth decreased significantly following each burn in B and MB, and recovered to pretreatment levels within one or two years as leaves dropped from trees each fall. Percent canopy openness at the drift fence-level was marginally (P < 0.1) greater in MB than C or M and differed among years; we did not detect an interaction effect.

**Table 2**. Results of mixed-model ANOVA comparing treatment, year, and treatment x year interaction effects on select forest structural features at Green River Game Land, Polk County, NC. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and sampling schedules.

Habitat variable	Mixed-1	Mixed-model ANOVA results							
	P <sub>trt</sub>	Pyr	P <sub>trtxyr</sub>	Treatment differences					
Live tree density	0.002	< 0.001	< 0.001	C <sup>a</sup> M <sup>a</sup> B <sup>a</sup> MB <sup>b</sup>					
Live tree basal area	0.033	0.010	< 0.001	Ca Ma Bab MBb					
Snag density	0.023	< 0.001	0.025	Ca Ma Bab MBb					
Shrub stem density	0.015	< 0.001	< 0.001	Cab Ma Bab MBb					
Leaf litter depth	< 0.001	< 0.001	< 0.001	C <sup>a</sup> M <sup>a</sup> B <sup>b</sup> MB <sup>c</sup>					

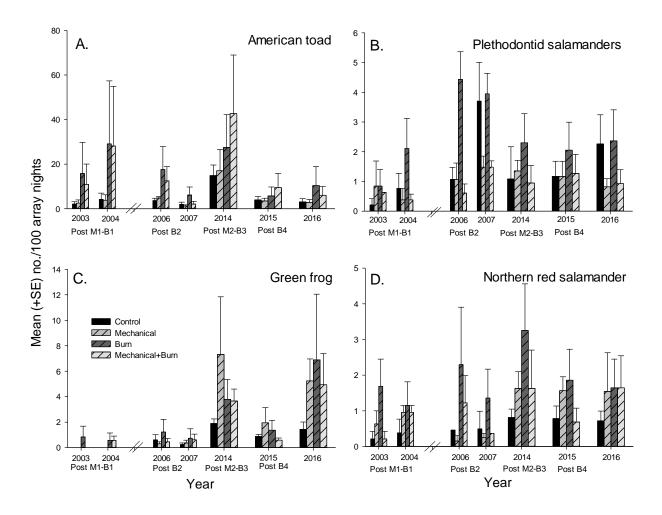
**Fig. 4**. Mean (±SE) live tree density and basal area, snag density, shrub stem density, and leaf litter depth in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, North Carolina, USA, 2001–2016.



### Reptiles and amphibians

We captured 3,299 individuals (60 recaptures; 1.5%) of 15 amphibian species, and 1,307 individuals (127 recaptures; 9.7%) of 20 reptile species (Table 3). Frogs and toads represented 53.8% of total first-captures followed by lizards (22.8%), salamanders (17.9%), snakes (5.4%), and turtles (0.2%). Only six species were sufficiently common ( $\geq$  200 individuals) for statistical analyses: five-lined skink (*Plestiodon fasciatus*), eastern fence lizard (*Sceloporus undulatus*), American toad (Anaxyrus americanus), green frog (Lithobates clamitans), plethodontid salamanders, and northern red salamander (Pseudotriton ruber). Species richness of amphibians, reptiles, and total herpetofauna differed among years but not among treatments; no treatment × year interaction effect was detected for total species richness (Table 3). Total species richness differed among treatments only in 2003 and 2004, and changed over time in all treatments and controls. In 2003 and 2004, total species richness was lower in C than B or MB. Within treatments, total species richness was lower in 2003 and 2004 than all subsequent years in C and M, all subsequent years except 2015 in B, and all subsequent years except 2006, 2015, and 2016 in MB. Capture rates of American toads, green frogs, plethodontid salamanders, and northern red salamanders differed among years but not among treatments; no treatment × year interaction effect were detected (Table 3; Fig. 5). Capture rates of five-lined skinks (44% of all lizard captures) differed among years, and were greater in MB than B or C (Table 3; Fig. 6a). Eastern fence lizard (44% of all lizard captures) capture rates also differed among years, and were greater in MB than C; no treatment × year effect was detected for either species (Table 3; Fig. 6b). No snake species was sufficiently common for statistical analyses. Juvenile American toad and green frog capture rates differed among years but not among treatments, and no treatment × year interaction was detected (Table 4). Juvenile five-lined skink capture rate differed among years but not among treatments, and no treatment × year interaction was detected (Table 4; Fig. 7a). Juvenile eastern fence lizard capture rate differed among years and was marginally greater in MB than C; no treatment × year interaction was detected (Table 4; Fig 7b). The capture rate of juvenile eastern fence lizards changed over time in MB, and differed among treatments in 2014, 2015, and 2016; capture rate was greater in MB than all other treatments in 2014, and marginally greater in 2015 and 2016. Within MB, capture rate of juveniles was lower in 2003 and 2004 than 2014, 2015, and 2016 (P = 0.005), and lower in 2006 and 2007 than 2014 and 2015 ( $P \le 0.0207$ ).

**Fig. 5**. Mean (<u>+</u>SE) number of first-captured American toads (A), plethodontid salamanders (B), green frogs (C), and northern red salamanders (D) per 100 array nights in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, North Carolina, 2003–2016.



**Table 3**. Number first-captured (and recaptured) reptiles and amphibians (all years and units combined; 20,772 array nights), and results of mixed-model analysis of variance (ANOVA) comparing treatment, year, and treatment × year interaction effects on first-captures of common (≥200 first-captures) species/100 array-nights, and species richness at Green River Game Land, Polk County, NC, 2003–2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and trapping schedules. In the treatment differences column, treatments with different superscripted letters within a row differ at the  $P \le 0.05$  level.

	Total Captured	Mixed-model ANOVA results				
		P <sub>trt</sub>	$\mathbf{P}_{\mathbf{yr}}$	P <sub>trtxyr</sub>	Treatment differences	
Amphibians	3,299 (60)					
Frogs and toads, Anura	2,476 (49)					
American toad (Anaxyrus americanus)	2,040 (42)					
American bullfrog (Lithobates catesbeianus)	12(0)					
Narrowmouth toad (Gastrophryne carolinensis)	1 (0)					
Gray treefrog (Hyla versicolor chrysocelis)	3 (0)					
Green frog (L. clamitans)	395 (7)	0.891	< 0.001	0.795		
Pickerel frog (L. palustris)	13 (0)					
Wood frog (L. sylvaticus)	12(0)					
Salamanders, Caudata	823 (11)					
Blue Ridge two-lined salamander (Eurycea wilderae)	71 (1)					
Eastern newt (Nothophthalmus viridescens)	183 (1)					
Plethodontid salamanders <sup>1</sup>	326 (3)	0.206	0.003	0.710		
Northern red salamander (Pseudotriton ruber)	236 (6)	0.341	0.004	0.215		
Northern dusky salamander (Desmognathus fuscus)	1 (0)					
Seal salamander (D. monticola)	5 (0)					
Blackbelly salamander (D. quadramaculatus)	1 (0)					
Reptiles	1,307 (127)					
Lizards (Lacertilia)	1,050 (114)					
Green anole (Anolis carolinensis)	1 (0)					
Broad-headed skink (Plestiodon laticeps)	55 (9)					

Table 3 continued.

	Total	Mixed-model ANOVA results					
	Captured	P <sub>t-st</sub>	P <sub>trt</sub> P <sub>yr</sub> P <sub>trtxyr</sub> Treatment				
		- uri	2 yr	- urtxyr	differences		
Coal skink (P. anthracinus)	53 (0)						
Five-lined skink ( <i>P. fasciatus</i> )	464 (57)	0.025	< 0.001	0.363			
Eastern fence lizard (Sceloporus undulatus)	463 (48)	0.032	< 0.001	0.559	Ca Mab Bab MBb		
Ground skink (Scincella lateralis)	14 (0)						
Snakes, Serpentes	250 (11)						
Copperhead (Agkistrodon contortix)	14 (0)						
Eastern worm snake (Carphophis amoenus)	145 (7)						
Black racer (Coluber constrictor)	2(0)						
Timber rattlesnake (Crotalis horridus)	2(0)						
Ring-necked snake (Diadophis punctatus)	47 (3)						
Eastern hog-nosed snake (Heterodon platirhinos)	5 (0)						
Milk snake (Lampropeltis triangulum)	5 (0)						
Northern water snake (Nerodia sipedon)	3 (0)						
Corn snake (Pantherophis guttatus)	1 (0)						
Eastern rat snake (P. alleghaniensis)	3 (0)						
Redbelly snake (Storeria occipitomaculata)	3 (0)						
Common garter snake (Thamnophis sirtalis)	20(1)						
Turtles, Testudinides	7 (2)						
Snapping turtle (Chelydra serpentina)	1 (0)						
Eastern box turtle (Terrapene Carolina)	6 (2)						
Total amphibian richness <sup>2</sup>	14	0.319	< 0.001	0.058			
Total reptile richness	20	0.067	< 0.001	0.653			
Total richness <sup>b</sup>	34	0.597	< 0.001	0.015			

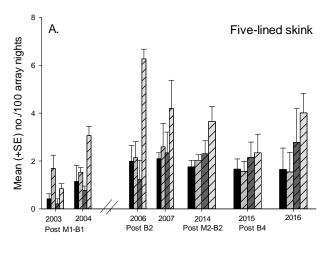
<sup>&</sup>lt;sup>1</sup> Includes white-spotted slimy salamander (*Plethodon glutinosis* complex) and southern gray-cheeked salamander (*P. jordani* complex) (91% and 9% of plethodontid salamanders captured, respectively, based on all plethodontid salamander captures 2006, 2007, 2014-2016 (n = 295, all positively identified). <sup>2</sup> Richness counts lump both plethodontid salamander species.

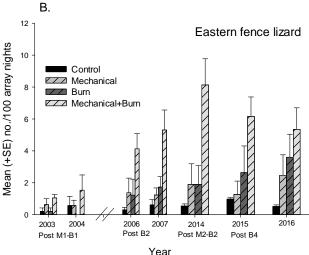
**Table 4.** Number of individual (first-captures) adults (A) and juveniles (J) of species having ≥200 captures (all years and units), and results of mixed-model analysis of variance (ANOVA) comparing treatment, year, and treatment × year interaction effects on adult or juvenile captures/100 array-nights at Green River Game Land, Polk County, NC, 2003–2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and trapping schedules. In the treatment differences column, treatments with different superscripted letters within a row differ at the  $P \le 0.05$  level.

	Age-class	Total		Mixed-m	odel AN	OVA results
			P <sub>trt</sub>	P <sub>yr</sub>	P <sub>trtxyr</sub>	Treatment differences
Five-lined skink (Plestiodon fasciatus)	A	276	0.004	0.001	0.411	C <sup>a</sup> M <sup>a</sup> B <sup>a</sup> MB <sup>b</sup>
	J	188	0.173	0.011	0.534	
Eastern fence lizard (Sceloporus undulatus)	A	404	0.033	< 0.001	0.736	Ca Mab Bab MBb
	J	59	0.092	0.003	0.005	
American toad (Anaxyrus americanus)	A	512	0.276	< 0.001	0.728	
	J	1,528	0.795	< 0.001	0.910	
Green frog (Lithobates clamitans)	A	24				
	J	371	0.904	< 0.001	0.732	
Plethodontid salamander <sup>1</sup>	A	287	0.244	0.001	0.827	
	J	39				
Northern red salamander (Pseudotriton ruber)	A	212	0.330	0.001	0.064	
	J	24				

Includes white-spotted slimy salamander (*Plethodon glutinosis* complex) and southern gray-cheeked salamander (*Plethodon jordani* complex) (91% and 9% of plethodontid salamanders captured, respectively, based on all plethodontid salamander captures 2006, 2007, 2014–2016 (n = 295, all positively identified).

**Fig. 6**. Mean (<u>+</u>SE) number of first-captured five-lined skinks (A), and eastern fence lizards (B) per 100 array nights in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, North Carolina, USA, 2003–2016.



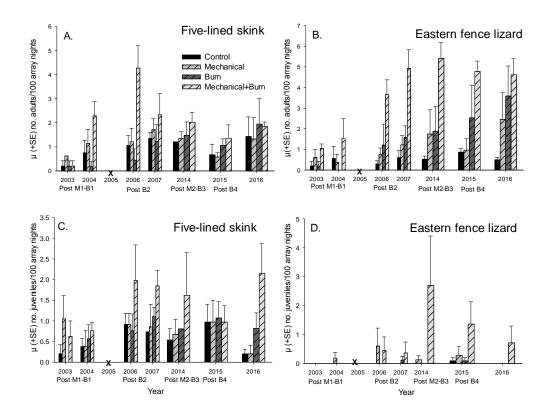


#### **Breeding Birds**

We detected 7,236 individuals of 56 breeding bird species during the 11 years sampled between 2001 and 2016. Among the 25 species meeting our criteria for analysis, relative density of 17 showed a response to treatments (Table 5). Several common species showed no response to treatments; some showed temporary decreases in density after burns, and several increased in MB, where the initial high-severity burn killed a high proportion of trees, creating an open canopy with dense shrubs.

Total bird density was greater in MB than C, M, or B (47.0%-149.5% greater than all other treatments, beginning in 2005) and differed among years; no treatment x year interaction was detected (Table 5; Fig. 8a). Total bird density was negatively associated with live tree density and positively associated with shrub stem density (Table 6). Species richness differed among years and was greater in MB than C and M; a treatment x year interaction was detected

**Fig. 7**. Mean (±SE) number of first-captured adult (A and B) and juvenile (C and D) fivelined skinks and Eastern fence lizards per 100 array nights in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, USA, 2003-2016.



(Table 5; Fig. 8b). Species richness changed over time in M, B, and MB, and differed between at least some treatments every year, starting three breeding seasons after initial burns (2005). Within M, richness was greater after the second mechanical treatment (2012) than three and four years after the initial mechanical treatment (2004 or 2005). Within B, species richness was intermittently greater than pretreatment, beginning four breeding seasons after the second burn (2009, 2014, and 2016). Within MB, species richness was consistently, significantly greater (by 44.7% - 70.2%) than pretreatment beginning three breeding seasons after the initial high-severity burn (2005). Species richness was significantly greater (by 41.8% -119.4%) in MB than C or M most years beginning in 2005, but significantly greater than B only in 2006 and 2012. Species richness was greater in B than C and M only in 2014, and did not differ between M and C in any year (Fig. 8b). Total species richness was negatively associated with tree density and positively associated with snag density and leaf litter depth (Table 6).

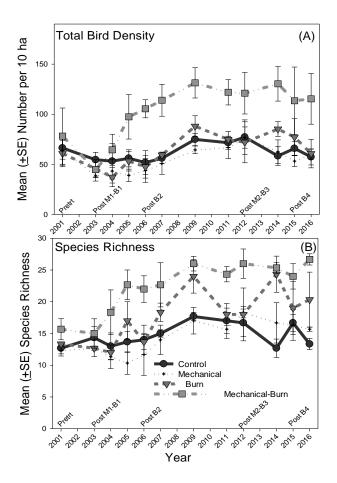
**Table 5.** Total number of individual bird detections (all years, units, and point counts) and results of mixed-model ANOVA comparing treatment<sup>1</sup>, year, and treatment x year interaction effects on breeding bird species richness and total density (no/10 ha), and density by species<sup>2</sup> (if  $\geq$ 75 observations) and nesting guilds at Green River Game Land, Polk County, NC, 2001-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and bird sampling schedules. In the treatment differences column, treatments with different superscripted letters within a row differ at the  $P \leq 0.05$  level.

	Total Observations		Mixed-model ANOVA results					
	Observations	P <sub>trt</sub>	P <sub>yr</sub>	P <sub>trtxyr</sub>	Treatment differences			
Tree-nester	2134	0.1165	0.0020	0.9709				
Acadian flycatcher	33							
American crow	31							
American redstart	7							
Blue-gray gnatcatcher	318	0.459	< 0.001	0.171				
Brown-headed cowbird	102	0.162	< 0.001	0.265				
Blue jay	94	0.737	0.514	0.383				
Black-throated green warbler	257	0.041	< 0.001	0.616	Ca Mab Bab MBb			
Broad-winged hawk	14							
Cedar waxwing	94	0.039	< 0.001	0.1341	Ca Mab Bab MBb			
Common grackle	4							
Coopers Hawk	2							
Eastern wood-pewee	212	0.002	< 0.001	0.124	Ca Mab Bbc MBc			
Northern parula	3							
Pine warbler	76	0.020	0.003	0.702	Ca Ma Bab MBb			
Red-eyed vireo	512	0.484	0.044	0.058				
Scarlet tanager	203	0.580	0.373	0.387				
Sharp-shinned hawk	2							
Summer tanager	6							
Yellow-billed cuckoo	5							
Yellow-throated vireo	17							
Yellow-throated warbler	59							
Shrub-nester	2234	0.001	< 0.001	0.057	Ca Ma Ba MBb			
American goldfinch	142	0.019	0.139	0.345				
American robin	10							
Blue-headed vireo	506	0.277	0.023	0.561				
Brown thrasher	7							
Chipping sparrow	29							
Eastern towhee	378	0.007	< 0.001	< 0.001	Ca Ma Ba MBb			
Hooded warbler	617	0.033	< 0.001	0.175				
Indigo bunting	206	0.002	< 0.001	0.003	Ca Ma Ba MBb			
Mourning dove	84	0.010	< 0.001	0.026	Ca Ma Ba MBb			
Northern cardinal	59							

Table 5 continued.

	Total Observations	Mixed-model ANOVA results					
		P <sub>trt</sub>	$\mathbf{P_{yr}}$	P <sub>trtxyr</sub>	Treatment differences		
Prairie warbler	32						
Ruby-throated hummingbird	192	0.451	0.549	0.728			
Swainson's warbler	4						
Wood thrush	44						
Yellow-breasted chat	7						
Cavity-nester	1795	0.001	0.275	0.868	Cab Mb Ba MBc		
Barred owl	1						
Carolina chickadee	265	0.043	0.063	< 0.001	Cab Ma Bab MBb		
Carolina wren	222	0.061	< 0.001	0.006			
Chimney swift	1						
Downy woodpecker	145	0.042	0.031	0.859	Cab Ma Bab MBb		
Eastern bluebird	88	0.006	0.007	0.263	Ca Ma Ba MBb		
Great-crested flycatcher	8						
Hairy woodpecker	29						
Pileated woodpecker	55						
Red-bellied woodpecker	58						
Red-headed woodpecker	12						
Tufted titmouse	452	0.043	0.056	0.990	Cab Ma Bab MBb		
White-breasted nuthatch	395	0.013	0.020	0.319	Ca Ma Bab MBb		
Yellow-shafted flicker	18						
Ground-nester	923	0.013	< 0.001	0.105	Ca Ma Bab MBb		
Black-and-white warbler	216	0.198	< 0.001	0.459			
Kentucky warbler	1						
Ovenbird	347	0.032	< 0.001	0.741	Cab Ma Bab MBb		
Worm-eating warbler	345	0.056	< 0.001	0.2075	Ca Map Bap MBp		
Wild turkey	14						
Other	31						
Eastern phoebe	31						
Total	7236	0.006	< 0.001	0.749	Ca Ma Ba MBb		
Species Richness	56	0.009	< 0.001	0.015	Ca Ma Bab MBb		

**Fig. 8**. Mean (±SE) total (A) density, and (B) species richness of breeding birds in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, NC, 2001–2016.



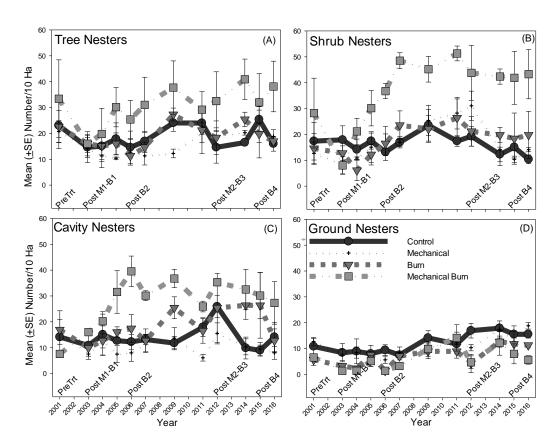
Relative density of tree-nesters differed among years but not treatments; no treatment x year interaction was detected, but trends suggested greater density in MB (Table 5; Fig. 9a). Tree-nester density was positively associated with shrub stem density (Table 6). Shrub-nester density was greater in MB than in other treatments, and density differed among years; a marginal treatment x year interaction effect was detected (p=0.0572) (Table 5; Fig. 9b). Shrub-nester density changed over time in M, B, and MB, and differed among some treatments each year starting in 2003. In M, shrub-nester density was greater in 2009 and 2011, several years after the first understory removal, and also immediately after the second understory removal (2012) compared to 2003 (a year after the first mechanical understory removal), but never differed significantly from pretreatment levels. Within B, shrub-nester density was lower in 2004 than several subsequent years (2007, 2009, 2011, 2012, 2014, 2015, and 2016) but never differed from pretreatment levels. In MB, shrub-nester density decreased relative to pretreatment levels immediately following the initial, high-severity burn (2003), but significantly increased by 2004, and remained greater than 2003 levels – nearly double that of other treatments, most years - for the duration of the study and throughout three subsequent burns (Figure 9b). Shrub-nester density was lower in B than in MB or C in 2004, and greater in MB than C or M in 2007, 2014,

2015, and 2016. Shrub-nester density was negatively associated with live tree density, and positively associated with shrub stem density (Table 6). Cavity-nester density was greater in MB than in other treatments, and density in B was greater than in M; no year or treatment x year effects were detected (Table 5; Fig. 9c). Cavity-nester density was negatively associated with live tree density and positively associated with snag density (Table 6). Ground-nester density differed among years, and was lower in MB than C or M; no treatment x year interaction effect was detected (Table 5; Fig. 9d) but trends suggested decreased density immediately after burning in both B and MB, followed by increases within two breeding seasons (Fig. 9d). Ground-nester density was positively associated with leaf litter depth and negatively associated with snag density (Table 6).

**Table 6**. Results of stepwise multiple regression of total bird density, bird density within nesting guilds, and species richness with habitat features including live tree ( $\geq 10$  cm dbh) and snag ( $\geq 10$  cm dbh) density, live tree basal area, shrub (woody stems  $\geq 1.4$  m ht and < 10 cm dbh) stem density, and leaf litter depth, Green River Game Land, Polk county, NC. Negative and positive relationships are indicated by - or + signs, respectively following habitat variable.

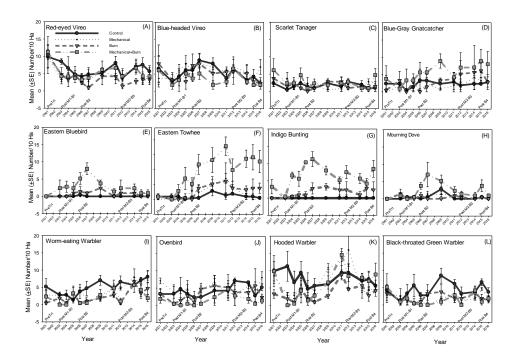
Nest guild	Habitat variable		Mode	l summary	y	
		Parameter	F	P	$\mathbf{r}^2$	RMSE
		Estimate (SE)				
Cavity-nester	Tree density (-)	-0.036 (0.007)	34.5	< 0.001	0.269	
	Snag density (+)	0.074 (0.021)	12.4	< 0.001	0.086	
	Total model		25.6	< 0.001	0.355	9.704
Ground-nester	Litter depth (+)	1.505 (0.246)	37.7	< 0.001	0.341	
	Snag density (-)	0.029 (0.012)	6.1	0.016	0.040	
	Total model		28.6	< 0.001	0.381	4.836
Shrub-nester	Tree density (-)	-0.042 (0.009)	66.5	< 0.001	0.4142	
	Shrub stem density (+)	0.002 (0.009)	9.0	0.004	0.051	
	Total model		40.5	< 0.001	0.465	10.184
Tree-nester	Shrub stem density (+)	0.002 (0.000)	13.6	< 0.001	0.126	
	Total model		13.6	< 0.001	0.126	10.969
Total density	Tree density (-)	-0.098 (0.022)	56.4	< 0.001	0.375	
	Shrub stem density (+)	0.003 (0.001)	4.7	0.033	0.030	
	Total model		31.7	< 0.001	0.405	24.664
Species richness	Tree density (-)	023 (0.003)	84.2	< 0.001	0.472	
	Snag density (+)	0.029 (0.009)	6.1	0.015	0.033	
	Litter depth (+)	0.407 (0.199)	4.2	0.043	0.022	
	Total model		34.1	< 0.001	0.527	3.655

**Fig. 9**. Mean (±SE) total density of breeding birds in (A) tree-, (B) shrub-, (C) cavity-, and (D) ground-nesting guilds in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, North Carolina, USA, 2001–2016.



Densities of many common species including red-eyed vireos (*Vireo olivaceus*), blueheaded vireos (*V. solitarius*), scarlet tanagers (*Piranga olivacea*), and blue-gray gnatcatchers (*Polioptila caerulea*) did not differ among treatments, and no treatment x year interaction effects were detected (Table 5; Fig. 10). Carolina wren density did not differ among treatments; a treatment x year effect was detected, but differences within treatments or years did not appear to be biologically meaningful (Table 5). Densities of other species were lower in MB than M or C. Black-throated green warbler (*S. virens*) and (marginally) worm-eating warbler (*Helmitheros vermivorus*) (p=0.0556) densities were lower in MB than C (Table 5; Fig. 10). Ovenbird (*Seiurus aurocapillus*) density was lower in MB than in M. Hooded warbler (*S. citrina*) density did not differ between MB and other treatments, but density was lower in B than C. No treatment x year interaction effects were detected for these species (Table 5; Fig. 10).

**Fig. 10**. Mean (±SE) total density of (A) red-eyed vireo; (B) blue-headed vireo; (C) scarlet tanager; (D) blue-gray gnatcatcher; (E) eastern bluebirds; (F) indigo buntings; (G) eastern towhees; (H) Eastern wood-pewee; (I) worm-eating warbler; (J) ovenbird; (K) hooded warbler; (L) black-throated green warbler in repeated dormant-season fuel reduction treatments and controls at Green River Game Land, Polk County, North Carolina, USA, 2001–2016.



Several species associated with open, young forest conditions were more abundant in MB than any other treatment. Eastern bluebird (Sialia sialis) density differed among years and were more abundant in MB than C, M, or B; no treatment x year interaction effect was detected (Table 5; Fig. 10). Eastern towhee (*Pipilo erythrophthalmus*) density differed among years and was greater MB than C, M, or B; a treatment x year interaction effect was detected (Table 5; Fig. 10). Eastern towhee density did not differ among C, M, or B within any sampled year. In M, eastern towhee density did not differ before and after either of the two mechanical understory removal treatments, but generally increased beginning five breeding seasons after the first removal treatment. In MB, eastern towhee density was greater beginning in 2005, compared to pretreatment or immediately after the initial, high-severity burns. Density was greater in MB than C all years, beginning in 2005, greater in M most years (2005, 2007, 2014, 2015, and 2016), but greater than B only in three years (2005, 2006, and 2011). In B, density was lower immediately after the first two burns (2003 and 2006) than several other years (2007, 2009, 2012, 2014, and 2016) but did not differ from pretreatment in any year. Indigo bunting (Passerina cyanea) density differed among years and was greater in MB than C, M, or B; a treatment x year interaction effect was detected (Table 5; Fig. 10). Indigo bunting density was consistently greater in MB than C, M, or B in all years except 2003, and did not differ between C and M in any year (Fig. 10). In B, indigo bunting density was greater two (2007) and four (2009) breeding seasons after the second burn, than pretreatment or immediately after initial

burns, but did not significantly differ from pretreatment in other years. In MB, indigo bunting density increased beginning two breeding seasons after initial high-severity burns (2004), and remained greater for the duration of the study period. Mourning dove (*Zenaida macroura*) density differed among years and was greater in MB than C, M, or B; a treatment x year interaction effect was detected (Table 5; Fig. 10). In MB, mourning dove density was greater than pretreatment in several, but not all years, beginning after the second burn (2006). Mourning dove density was greater in MB than C, M, or B during some years, beginning in 2006. Mourning dove density was lower in B than C only in 2009, and did not differ between M and C in any year.

Density of several additional species was greater in MB and (or) B than in some other treatments, with no treatment x year interactions detected (Table 5). Cedar waxwings (Bombycilla cedrorum) and American goldfinch (Carduelis tristis) densities were greater in MB than C. Eastern wood-pewee (Contopus virens) density was greater in MB than C or M, and greater in B than C. White-breasted nuthatch (Sitta carolinensis) and pine warbler (S. pinus) densities were greater in MB than C or M, and downy woodpeckers (Picoides pubescens) and eastern tufted titmice (Baelolophus bicolor) densities were greater in MB than in M. Carolina chickadee (Poecile carolinensis) density was greater in MB than in M; a treatment x year effect was detected, but differences within treatments or years did not appear to be biologically meaningful.

## **Pollinating Insects**

We captured 5,520 flower visitors (2014-2016) (Table 7). Bees, including 20 genera, were most common (53.9% of total captures); Lasinglossum spp. (40.3%) and Augochlora pura Say (51%) dominated total bee captures. Flower-visiting wasps (38.7%), syrphid flies (5.5%) and butterflies (1.9%) were also commonly captured. Among wasps, Vespula spp. (39.1%), Pompilidae (31.3%), and Tiphiidae (17.4%) were most common. On the forest floor, bee abundance (driven by *Lasioglossum* spp.) was greater in MB than M or C ( $\chi^2$ = 16.17, df = 3, P < 0.0001) (Table 8), and greater in B than C. Captures of both *Lasioglossum* ( $\chi^2$ = 16.01, df = 3, P = 0.0002) and *Bombus* ( $\chi^2$ = 10.77, df = 3, P = 0.007) bees were greater in MB than C (Table 8). Forest floor bee diversity (df = 3, 34; F = 11.41, P < 0.0001) and genera richness (df = 3, 34; F = 11.41, P < 0.0001) 9.63, P < 0.0001) was also greater in MB than M or C (Table 9). The abundance of common wasp families did not differ among treatments. Syrphid fly abundance on the forest floor was greater in M, B, and MB compared to C ( $\chi^2 = 14.52$ , df = 3, P = 0.0006). No treatment effects were detected for any commonly captured insects in midstory traps. However, we detected numerous differences between the forest floor and midstory strata. Total bee abundance was greater in midstory- than forest floor traps (z = 2.55, P = 0.01); this trend was driven by Augochlora pura (z = 5.09, P < 0.0001), the most commonly captured bee species (Table 8). Among common wasp families, Pompilidae (z = 6.60, P < 0.0001) captures were greater on the forest floor, whereas Vespula and Dolichovespula maculata L. captures were greater in the midstory (df = 69, t = 6.65, P < 0.0001; z = 2.56, P = 0.05, respectively) (Table 8). Captures of the most common butterfly family, Hesperiidae, were greater on the forest floor than the midstory (df= 69, t= 1.66, P = 0.01).

**Table 7**. Total number of all flower-visiting invertebrate species (forest floor and midstory captures combined) captured with colored pan traps at Green River Game Land, Polk County, NC, 2014-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules.

Order	Family	Genus/Species	Total Captured	В	C	M	MB
Hymenoptera (Bees)	Halictidae	Agapostemon virescens	4	3	0	0	1
		Augochlorella aurata	16	4	0	3	9
		Augochlora pura	1532	433	489	234	376
		Augochloropsis metallica	1	1	0	0	0
		Halictus confusus	1	0	1	0	0
		Halictus poeyi/lignatus	7	3	0	0	4
		Lasioglossum spp.	1198	368	113	201	516
		Sphecodes sp.	1	1	0	0	0
	Apidae	Anthophora abrupta	2	0	1	0	1
		Apis mellifera	7	3	1	0	3
		Bombus spp. (B. bimaculatus, griseocollis, impatiens, perplexus)	133	47	8	20	58
		Ceratina spp. (C. calcarata, dupla, strenua)	44	13	2	12	17
		Holcopasites calliopsidis	1	0	0	0	1
		Melissodes bimaculata	2	0	0	0	2
		Melissodes denticulata	1	0	0	0	1
		Svastra atripes	1	0	0	0	1
		Xylocopa virginica	2	1	0	0	1
	Megachilidae	Hoplitis simplex	1	0	0	0	1
		Megachile mendica	2	1	0	0	1
		Megachile mucida	1	1	0	0	0
		Megachile paralella	1	0	0	0	1
		Megachile relativa	1	0	0	0	1
	Andrenidae	Andrena cressonii/macoupinensis	2	0	0	1	1
		Andrena sp.	2	1	0	0	1
		Calliopsis andreniformis	4	1	0	0	3
	Colletidae	Hylaeus affinis/modestus	1	1	0	0	0
		Hylaeus annulatus	3	0	0	0	3
Hymenoptera (Wasps)	Chrysididae		46	9	16	13	8
	Crabronidae		30	8	8	8	6
	Mutillidae		79	9	34	24	12
	Pelicinidae	Pelicinus polyturator	2	1	0	0	1

Table 7 continued.

Order	Family	Genus/Species	Total Captured	В	C	M	MB
	Pompilidae		666	169	181	189	127
	Sphecidae		40	8	4	4	24
	Tiphiidae		370	131	56	77	106
	Vespidae	Dolichovespula maculata	54	20	6	18	10
		Eumeninae	5	1	1	0	3
		Polistes spp.	4	2	0	0	2
		Vespula squamosa & flavopilosa	833	190	192	209	242
Diptera	Bombyliidae		1	0	0	0	1
	Conopidae		3	1	1	0	1
	Syrphidae		301	80	42	73	106
Lepidoptera	Hesperiidae		103	22	9	26	46
	Lycaenidae		7	1	0	1	5
	Papilionidae		5	2	0	1	2

**Table 8**. Mean number ( $\pm$  SE) of common genera/species or families of flower visitors captured with colored pan traps from the forest floor and midstory at Green River Game Land, Polk County, NC, 2014-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules. The last two rows are the mean numbers ( $\pm$  SE) of common genera/species or families of flower visitors captured from all 12 treatment units combined. Treatments with different superscripted letters within a column differ at the  $P \le 0.05$  level.

	TB	Lg	Ap	<i>B</i> b	Pp	Vs	Dm	Sy	Hp
Forest Floor									
С	0.8 (0.2) <sup>c</sup>	$0.4 (0.2)^{b}$	0.4 (0.1)	$0.02 (0.02)^{b}$	2.7 (0.3)	1.2 (0.4)	0.04 (0.02)	$0.1 (0.06)^{b}$	0.1 (0.03)
M	2.8 (1.0) <sup>bc</sup>	$2.1~(0.8)^{ab}$	0.4 (0.1)	$0.1 (0.05)^{ab}$	2.9 (0.5)	1.4 (0.8)	0.2 (0.09)	1.0 (0.3) <sup>a</sup>	0.3 (0.1)
В	5.1 (2.0) <sup>ab</sup>	3.5 (1.3) <sup>a</sup>	0.9 (0.5)	$0.3 (0.2)^{ab}$	2.5 (0.3)	1.0 (0.4)	0.05 (0.02)	$0.9(0.3)^{a}$	0.3 (0.09)
MB	7.6 (1.9) <sup>a</sup>	5.4 (1.5) <sup>a</sup>	1.1 (0.4)	0.4 (0.1) <sup>a</sup>	1.9 (0.7)	1.4 (0.7)	0.07 (0.07)	1.4 (0.4) <sup>a</sup>	0.6 (0.2)
Midstory									
C	8.0 (2.5)	1.2 (0.3)	6.6 (2.2)	0.1 (0.05)	0.3 (0.08)	2.1 (0.4)	0.09 (0.06)	0.5 (0.1)	0.04 (0.02)
M	4.4 (0.9)	1.1 (0.3)	3.0 (0.8)	0.2 (0.08)	0.4 (0.08)	2.3 (0.2)	0.2 (0.1)	0.3 (0.07)	0.1 (0.06)
В	10.0 (3.3)	3.4 (2.1)	5.7 (1.7)	0.8 (0.7)	0.4 (0.09)	2.8 (0.6)	0.3 (0.1)	0.6 (0.2)	0.2 (0.09)
MB	8.4 (1.6)	3.0 (1.0)	4.4 (1.4)	0.8 (0.3)	0.3 (0.04)	3.0 (0.8)	0.1 (0.05)	0.5 (0.2)	0.2 (0.1)
Forest Floor	4.1 (0.8) <sup>b</sup>	2.9 (0.6)	$0.7 (0.2)^{b}$	0.2 (0.06)	2.5 (0.2) <sup>a</sup>	$1.2 (0.3)^{b}$	$0.08(0.03)^{b}$	0.8 (0.2)	$0.3 (0.07)^{b}$
Midstory	7.7 (1.1) <sup>a</sup>	2.2 (0.6)	$4.9(0.8)^{a}$	0.5 (0.2)	$0.4 (0.04)^{b}$	$2.5(0.3)^{a}$	$0.2 (0.05)^{a}$	0.5 (0.07)	0.1 (0.04) <sup>a</sup>

TB=total bees, Lg=*Lasioglossum* spp., Ap=*Augochlora pura*, Bb=*Bombus* spp., Pp=Pompilidae, Vs=*Vespula* spp., Dm=*Dolichovespula maculata*, Sy=Syrphidae, Hp=Hesperiidae.

**Table 9.** Mean ( $\pm$ SE) Shannon-Weiner and genera richness of bees captured with colored pan traps at Green River Game Land, Polk County, NC, 2014-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules.

Trap Location	Shannon-Weiner	Genera
	Index	Richness
Forest floor		
Control (C)	0.026 (0.02)	0.4 (0.2)
Mechanical (M)	0.177 (0.02)	0.9 (0.2)
Burn (B)	0.314 (0.09)	1.3 (0.5)
Mechanical+Burn (MB)	0.475 (0.09)	1.7 (0.4)
Midstory		
Control (C)	0.213 (0.04)	1.3 (0.04)
Mechanical (M)	0.234 (0.05)	1.7 (0.3)
Burn (B)	0.279 (0.07)	2.2(0.4)
Mechanical+Burn (MB)	0.366 (0.05)	1.8 (0.3)
Forest Floor Total	0.273 (0.03)	1.1 (0.2)
Midstory Total	0.250 (0.04)	1.7 (0.2)

#### **Beetles**

We captured 7,037 beetles comprising 62 families and at least 210 species (2014-2016) (Table 10). Curculionidae (20% of total captures; subfamily Scolytinae comprising more than half) the most commonly captured family, followed by Nitidulidae (12.9%), Elateridae (10.7%), Scarabaeidae (10%), Staphylinidae (10%), and Mordellidae (9.1%). Fungivores (28.9% of total captures) were the most common adult feeding guild, followed by phytophagous/fungivores (12.8%), phytophagous (11.9%), and sap/sugar feeders (11.5%). Capture rate of several Coleoptera families differed among the treatments (Table 11). Nitidulidae captures were greater in C than MB or B, whereas Mordellidae captures were greater in B than M or C (Table 11). However, no differences among treatments in capture rate were detected for any common species (n ≥ 50 individuals). Among adult feeding guilds (Table 12). fungivores and phytophagous/predator beetles captures were greater in C than MB. Phytophagous and coprophagous beetle captures were greater in B than M. Shannon-Weiner diversity indices of focal families (Carabidae, Cerambycidae, Cleridae, and Scarabaeidae) did not differ among treatments (Table 13). Species richness of Scarabaeidae differed among treatments, but no differences were detected for the other focal families (Table 13).

**Table 10**. Total number of beetles captured at Green River Game Land, Polk County, NC, 2014-2016. Feeding guilds are based on adult behaviors. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	С	Total Captured	Feeding Guild
Aderidae	•			7	7	1	2	17	Unknown
			Zonantes fasciatus	0	0	1	0	1	
Agyrtidae			Necrophilus pettiti	0	0	2	0	2	Scavenger
Anthicidae				1	1	1	1	4	Scavenger
Anthribidae				4	2	2	0	8	Fung/Sapro
			Eurymycter fasciatus	1	0	1	0	2	
Attelabidae				0	0	0	2	2	Phytophagous
			Synolabus bipustulatus	1	0	2	3	6	
			Temnocerus aeratus	0	0	1	0	1	
Brentidae				1	0	1	1	3	Phyto/Sapro
	Apioninae			0	0	1	0	1	
Buprestidae				6	6	0	0	12	Phytophagous
			Agrilus sp.	0	2	1	0	3	
			Brachys sp.	1	0	0	0	1	
			Chrysobothris sp.	7	1	2	2	12	
			Actenodes sp.	2	0	0	0	2	
			Acmaeodera sp.	4	1	0	0	5	
Cantharidae			Brachys sp.	2	3	1	0	6	Phyto/Pred
	Malthininae	Malthinini	Malthinus sp.	0	0	0	1	1	
	Chauliognathinae		Trypherus sp.	3	0	0	3	6	
	Silinae		Tytthonyx sp.	0	0	1	0	1	
Carabidae	Carabinae	Carabini	Calosoma sp.	1	0	0	0	1	Predator
			Carabus sp.	6	1	3	4	14	
	Cicindelinae	Cicindelini	Cicindela sexguttata	11	11	8	2	32	
			Cicindela unipunctata	2	13	5	0	20	
	Harpalinae	Chlaeniini	Chlaenius sp.	0	0	0	1	1	
		Galeritini	Galerita sp.	7	0	4	3	14	
		Harpalini	Anisodactylus sp.	1	0	0	0	1	
		Lebiini	Cymindis sp.	0	0	0	1	1	
			Lebia sp.	2	3	5	7	17	
			Nemotarsus sp.	0	1	0	0	1	

Table 10 continued.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	C	Total Captured	Feeding Guild
		Licinini	Dicaelus sp.	5	2	1	2	10	
		Platynini	Agonum sp.	0	0	1	1	2	
		·	Platynus sp.	0	0	1	1	2	
		Pterostichini	Cyclotrachelus sp.	0	1	0	0	1	
			Pterostichus sp.	5	9	6	17	37	
	Nebriinae	Notiophilini	Notiophilus sp.	0	2	0	0	2	
	Rhysodinae		Clinidium sp.	2	2	0	1	5	
	Scaritinae	Pasimachini	Pasimachus sp.	2	0	0	0	2	
	Trechinae	Bembidiini	Mioptachys flavicauda	0	1	0	0	1	
			Miptachys sp.	0	1	0	0	1	
Cerambycidae	Cerambycinae	Clytini	Xylotrechus colonus	2	1	0	0	3	Phyto/Sapro
	·	Eburiini	Eburia quadrigeminata	1	0	0	0	1	•
		Elaphadiini	Parelaphidion incertum	1	0	0	0	1	
		Elaphidiini	Elaphidion mucronatum	0	2	0	0	2	
		Trachyderini	Purpuricenus paraxillaris	0	0	0	1	1	
	Lamiinae	Acanthocinini	Graphisurus fasciatus	1	0	0	0	1	
			Urgleptes querci	0	1	1	0	2	
		Acantoderini	Aegomorphus modestus	0	0	2	0	2	
		Monochamini	Microgoes oculatus	0	0	1	2	3	
		Pogonocherini	Ecyrus dasycerus	2	0	0	1	3	
	Lepturinae	Lepturini	Analeptura lineola	1	7	2	5	15	
			Brachyleptura rubrica	0	0	1	0	1	
			Grammoptera haematites	0	1	0	0	1	
			Strangalia famelica	0	0	0	1	1	
			Strangalia famelica famelica	0	0	1	0	1	
			Trachysida mutabilis	0	1	0	0	1	
			Typocerus velutinus	1	0	0	1	2	
		Rhagiini	Gaurotes cyanipennis	3	0	0	0	3	
			Metacmaeops vittata	6	2	6	9	23	
			Pidonia aurata	4	2	6	2	14	
Cerylonidae				0	1	1	0	2	Fungivore
Chrysomelidae				21	6	7	7	41	Phytophagous
	Cassidinae	Chalepini	Baliosus nervosus	0	0	0	1	1	
		_	Baliosus sp.	0	0	0	1	1	

Table 10 continued.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	С	Total Captured	Feeding Guild
-	Cryptocephalinae	Cryptocephalini	Cryptocephalus guttulatus	0	1	0	0	1	
	Galerucinae	Alticini	Altica sp.	0	0	0	1	1	
			Capraita circumdata	2	6	1	0	9	
			Capraita sp.	0	4	3	3	10	
			Capraita subvittata	0	0	0	2	2	
Ciidae				0	2	3	0	5	Fungivore
	Ciinae	Ciini	Cis sp.	1	0	0	0	1	
Clambidae			•	1	0	1	0	2	Fungivore
Cleridae	Clerinae		Enoclerus sp.	3	0	0	0	3	Predator
	Epiphloeinae		Madoniella dislocata	2	0	1	0	3	
	Hydnocerinae		Phyllobaenus pallipennis	16	14	17	3	50	
			Phyllobaenus sp.	0	1	3	0	4	
			Phyllobaenus unifasciatus	0	0	1	0	1	
	Neorthopleurinae		Neorthopleura thoracica	1	0	0	0	1	
	Peloniinae		Pelonium sp.	0	0	1	0	1	
			Chariessa pilosa	0	1	0	0	1	
	Tillinae		Cymatodera bicolor	1	0	1	0	2	
			Cymatodera sp.	0	1	0	0	1	
			Phyllobaenus pallipennis	0	0	1	0	1	
Coccinellidae				16	11	4	8	39	Predator
	Scymninae		Brachiacantha sp.	1	0	0	0	1	
Corylophidae			•	1	0	1	1	3	Fungivore
Cryptophagidae				1	3	6	8	18	Fungivore
Cupedidae			Tenomerga cinerea	5	12	0	1	18	Fung/Sapro
Curculionidae				214	165	93	91	563	Phyto/Fung
	Platypodinae			1	0	0	0	1	, ,
	Scolytinae			144	393	160	144	841	
Dermestidae	·	Anthrenini		3	0	0	0	3	Scavenger
			Cryptorhopalum sp.	2	0	0	0	2	
Elateridae				146	141	204	254	745	Sap/Sugar
	Dendrometrinae	Dendrometrini	Limonius basilaris	0	6	1	0	7	• •
			Limonius sp.	0	1	0	0	1	
	Lissominae		Drapetes sp.	1	0	0	0	1	
			Drapetes quadripustulatus	1	0	0	0	1	

Table 10 continued.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	C	Total Captured	Feeding Guild
Endomychidae	Lycoperdininae		Aphorista vittata	0	1	0	1	2	Fungivore
•			Mycetina perpulchra	2	2	1	4	9	-
			Mycetina sp.	0	4	1	2	7	
Erotylidae	Erotylinae	Tritomini	Triplax sp.	11	23	9	3	46	Fungivore
			Tritoma sp.	6	17	6	16	45	
Eucinetidae				1	1	3	4	9	Fungivore
			Eucinetus sp.	2	1	0	0	3	
Eucnemidae				13	16	12	11	52	Fungivore
	Melasinae			0	2	2	0	4	
	Macraulacinae	Macraulacini	Deltometopus amoenicornis	2	0	0	0	2	
	Melasinae	Melasini	Isorhipis obliqua	0	0	1	0	1	
		Melasini	Isorhipis sp.	0	1	0	0	1	
Geotrupidae	Bolboceratinae		Odonteus sp.	0	0	0	1	1	Fungivore
	Geotrupinae		Geotrupes sp.	1	3	8	3	15	Scavenger
Histeridae				9	26	12	6	53	Predator
	Saprininae			0	0	1	0	1	
Hydrophilidae				2	12	30	53	97	Phyto/Pred
	Sphaeridiinae	Megasternini	Cercyon sp.	0	0	0	3	3	
Laemophloeidae				1	2	1	0	4	Fung/Pred
Lampyridae				1	0	0	3	4	Unknown
	Lampyrinae	Lampyrinae	Lucidota sp.	1	1	0	5	7	
		Lucidotini	Photinus sp.	0	0	0	1	1	
		Photinini	Lucidota punctata	0	2	0	4	6	
			Phausis sp.	1	4	1	0	6	
	Photurinae		Photuris sp.	1	0	0	0	1	
Latridiidae				4	4	5	1	14	Fungivore
	Corticariinae		Melanophthalma sp.	0	0	0	1	1	
Leiodidae				2	0	7	7	16	Fungivore
	Choledinae			0	0	0	3	3	
	Leiodinae	Agathidiini		0	0	0	1	1	
		Agathidiini	Agathidium sp.	1	0	0	0	1	
Lucanidae	Lucaninae	Platycerini	Platycerus virescens	0	0	1	0	1	Sap/Sugar

Table 10 continued.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	C	Total Captured	Feeding Guild
Lycidae				0	1	0	0	1	Sap/Sugar
	Erotinae		Eropterus sp.	0	1	0	1	2	
	Erotinae		Erotides sp.	0	9	0	1	10	
			Plateros sp.	18	7	5	7	37	
Lymexylidae			Melittomma sericeum	2	1	0	0	3	Fungivore
Melandryidae				1	2	0	1	4	Fungivore
•	Melandryinae	Dircaeini	Dircaea liturata	4	5	1	0	10	-
	Melandryinae	Hypulini	Microtonus sericans	0	0	1	0	1	
Melyridae				0	0	1	0	1	Phtyo/Pred
	Malachiinae	Malachiini	Hypebaeus bicolor	1	0	0	0	1	•
	Malachiinae	Malachiini	Hypebaeus sp.	1	0	0	2	3	
Monotomidae				1	0	0	0	1	Phytophagous
Mordellidae				204	209	106	118	637	Phytophagous
Mycetophagidae				2	1	1	0	4	Fungivore
, ,			Litargus tetraspilotus	0	0	1	0	1	
Nitidulidae				45	103	261	503	912	Fungivore
Orsodacnidae				1	0	0	0	1	Phytophagous
Phalacridae				0	4	1	0	5	Fungivore
Passandridae			Catogenus rufus	0	0	1	0	1	Unknown
Ptilodactylidae				0	0	0	1	1	Fungivore
•	Ptilodactylinae		Ptilodactyla sp.	3	0	7	0	10	
Ptinidae	•			40	43	27	30	140	Scav/Sapro
	Anobiinae			0	0	1	0	1	•
	Anobiinae	Nicobiini	Trichodesma sp.	0	0	1	0	1	
	Dorcatominae		1	0	0	5	0	5	
	Xyletininae	Lasiodermini	Lasioderma sp.	0	1	0	0	1	
Pyrochroidae	·			0	1	0	0	1	Phytophagous
•	Pyrochroinae		Dendroides sp.	0	0	1	0	1	, 1
Salpingidae			1	0	0	0	1	1	Scavenger
Scarabaeidae	Aphodiinae			26	9	3	8	46	Phytophagous
	Cetoniinae	Trichiini	Trichiotinus sp.	0	1	0	0	1	7 1 0
	Melolonthidae	Sericini	Serica sp.	0	0	1	0	1	
	Melolonthidae		Phyllophaga sp.	0	0	1	0	1	
	Melolonthinae	Diplotaxini	Diplotaxis sp.	0	1	0	0	1	

Continued on next page

Table 10 continued.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	C	Total Captured	Feeding Guild
	Melolonthinae	Macrodactylini	Macrodactylus sp.	1	0	0	0	1	
	Melolonthinae	Melolonthini	Phyllophaga sp.	0	2	1	1	4	
	Melolonthinae	Sericini	Serica sp.	7	8	6	0	21	
	Melolonthinae			0	0	1	0	1	
	Melolonthinae		Phyllophaga sp.	0	1	3	1	5	
	Rutelinae	Anomalini	Anomala sp.	1	0	0	0	1	
Scarabaeidae	Scarabaeinae	Ateuchini	Ateuchus histeroides	10	26	7	1	44	Coprophagous
	Scarabaeinae	Ateuchini	Ateuchus sp.	0	1	0	0	1	
	Scarabaeinae	Canthonini	Canthon sp.	4	12	18	11	45	
	Scarabaeinae	Canthonini	Canthon viridis	103	120	71	78	372	
	Scarabaeinae	Canthonini	Deltochilum gibbosum	0	0	3	0	3	
	Scarabaeinae	Onthophagini	Onthophagus hecate	0	0	1	0	1	
	Scarabaeinae	Onthophagini	Onthophagus sp.	25	40	29	64	158	
Scirtidae				0	2	2	1	5	Phytophagous
			Sacodes sp.	0	1	0	0	1	
			Sacodes pulchella	0	1	0	0	1	
Scraptiidae				7	18	5	5	35	Phytophagous
•	Anaspidinae			0	0	3	0	3	
	Scraptiinae	Scraptiini	Canifa pallipes	0	0	0	1	1	
Silphidae			Nicrophorus orbicollis	4	2	10	6	22	Copro/Carrion
•			Necrophila americana	0	0	5	2	7	
			Nicrophorus tomentosus	0	0	0	1	1	
			Nicrophorus sp	19	13	17	32	81	
Sphindidae			•	0	1	0	0	1	Fungivore
Staphylinidae				77	211	184	226	698	Unknown
- ·	Scaphidiinae			0	1	1	0	2	
	Staphylininae	Staphylinini	Philonthus sp.	0	0	2	0	2	
	Staphylininae	Staphylinini	Platydracus sp.	3	1	1	1	6	
Tenebrionidae			1	59	35	16	12	122	Fung/Sapro
	Alleculinae			14	19	3	10	46	
	Alleculinae		Capnochroa fuliginosa	0	0	1	1	2	
	Alleculinae		Mycetochara sp.	1	0	0	0	1	
	Tenebrioninae	Pedinini	Alaetrinus minimus	0	1	0	0	1	
Tetratomidae				1	1	0	0	2	Fungivore
	Penthinae		Penthe obliquata	1	0	0	0	1	

Continued on next page

Table 10 continued.

Family	Subfamily	Tribe	Genus/Species	MB	В	M	C	Total Captured	Feeding Guild
Throscidae				66	106	62	100	334	Phyto/Fung
Trogidae			Trox sp.	1	1	0	0	2	Scavenger
Trogossitidae				0	1	2	0	3	Predator
	Peltinae		Thymalus marginicollis	1	0	1	0	2	
	Trogossitinae	Trogossitini	Tenebroides sp.	2	0	0	0	2	
Zopheridae	Colydiinae	Synchitini		1	0	0	0	1	Fung/Sapro
		Synchitini	Endeitoma granulata	0	0	1	0	1	

**Table 11**. Mean number ( $\pm$ SE) of beetles from common (n  $\geq$  25) families captured at Green River Game Land, Polk County, NC, 2014-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules. Families with an \* indicate a significant difference at P  $\leq$  0.05 and \*\* indicate a significant difference at P  $\leq$  0.1. Within each family or subfamily, treatments with different superscripted letters within a row are significantly different.

Family	MB	В	М	С	Test Statistics
Family			M		(df= 2, 3)
Carabidae	2.4 (0.6)	2.6 (0.5)	1.2 (0.3)	2.2 (0.5)	F= 0.11, <i>P</i> = 0.9
Cerambycidae	1.2 (0.3)	0.9 (0.3)	1.1 (0.3)	1.2 (0.3)	F= 0.28, <i>P</i> = 0.8
Chrysomelidae	1.3 (0.6)	0.9 (0.3)	0.6 (0.2)	0.8 (0.2)	F= 1.03, <i>P</i> = 0.4
Cleridae*	1.3 (0.6) <sup>ab</sup>	0.9 (0.3)ab	1.4 (0.3) <sup>a</sup>	0.2 (0.1) <sup>b</sup>	F= 6.74, <i>P</i> = 0.02
Coccinellidae	0.9 (0.3)	0.6 (0.4)	0.2 (0.1)	0.4 (0.1)	F= 1.19, <i>P</i> = 0.4
Curculionidae**	19.9 (3.4) <sup>ab</sup>	31.0 (7.2) <sup>a</sup>	14.1 (2.7) <sup>ab</sup>	13.1 (2.5) <sup>b</sup>	F= 4.03, <i>P</i> = 0.07
Elateridae	8.2 (1.4)	8.2 (1.4)	11.4 (1.4)	14.1 (1.3)	F= 2.40, <i>P</i> = 0.2
Erotylidae	0.9 (0.3)	2.2 (0.9)	0.8 (0.3)	1.1 (0.4)	F= 0.83, <i>P</i> = 0.5
Eucnemidae	0.8 (0.2)	1.0 (0.3)	0.8 (0.2)	0.6 (0.2)	F= 0.49, <i>P</i> = 0.7
Histeridae	0.5 (0.2)	1.4 (0.5)	0.7 (0.4)	0.3 (0.1)	F= 0.50, P= 0.7
Hydrophilidae*	0.1 (0.08) <sup>b</sup>	0.7 (0.3)ab	1.7 (0.7)ab	3.1 (0.9) <sup>a</sup>	F= 4.75, <i>P</i> = 0.05
Lampyridae	0.2 (0.1)	0.4 (0.2)	0.06 (0.06)	0.7 (0.3)	F= 1.03, <i>P</i> = 0.4
Lycidae	1.0 (0.5)	1.0 (0.5)	0.3 (0.1)	0.5 (0.1)	F= 0.30, <i>P</i> = 0.8
Mordellidae*	11.3 (2.0) <sup>ab</sup>	11.6 (2.0) <sup>a</sup>	5.9 (0.8) <sup>b</sup>	6.6 (1.2) <sup>b</sup>	F= 5.79, <i>P</i> = 0.03
Nitidulidae*	2.5 (0.6)°	5.7 (1.2) <sup>bc</sup>	14.5 (4.3) <sup>ab</sup>	27.9 (9.4) <sup>a</sup>	F= 9.95, <i>P</i> = 0.001
Ptinidae	2.2 (0.9)	2.4 (1.0)	1.9 (0.8)	1.7 (0.6)	F= 0.29, <i>P</i> = 0.8
Scarabaeidae**	9.8 (1.3) <sup>ab</sup>	12.2 (1.6) <sup>a</sup>	8.1 (1.9) <sup>b</sup>	9.1 (1.1) <sup>ab</sup>	F= 3.71, <i>P</i> = 0.08
Scraptiidae	0.4 (0.3)	1.0 (0.5)	0.4 (0.2)	0.3 (0.2)	F= 1.45, <i>P</i> = 0.3
Silphidae	1.3 (0.6)	0.8 (0.5)	1.8 (0.5)	2.2 (0.7)	F= 0.63, P= 0.6
Staphylinidae*	4.4 (0.7) <sup>b</sup>	11.8 (2.6) <sup>a</sup>	10.4 (2.7) <sup>ab</sup>	12.6 (2.9) <sup>a</sup>	F= 8.9, <i>P</i> = 0.01
Tenebrionidae	4.1 (1.7)	3.1 (1.0)	1.1 (0.3)	1.3 (0.2)	F= 0.94, <i>P</i> = 0.5
Throscidae	3.7 (1.7)	5.9 (2.3)	3.4 (1.2)	5.6 (2.3)	F= 0.70, <i>P</i> = 0.6

**Table 12**. Mean numbers ( $\pm$ SE) of beetles from various adult feeding guilds captured at Green River Game Land, Polk County, NC, 2014-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules. Genera/family with an \* indicate a significant difference at  $P \le 0.05$  and \*\* indicate a significant difference at  $P \le 0.1$ . Within each guild, treatments with different superscripted letters within a row are significantly different.

Feeding Guild	МВ	В	M	C	Test Statistics (df= 2,3)
Fungivores**	13.8 (2.8) <sup>b</sup>	32.7 (7.3) <sup>ab</sup>	26.9 (5.4) <sup>ab</sup>	39.6 (10.4) <sup>a</sup>	F= 4.16, <i>P</i> = 0.06
Phytophagous*	14.8 (2.5) <sup>ab</sup>	15.1 (0.2) <sup>a</sup>	8.2 (1.0) <sup>b</sup>	8.3 (1.2) <sup>ab</sup>	F= 4.69, <i>P</i> = 0.05
Phytophagous/Predator**	0.5 (0.2) <sup>b</sup>	0.8 (0.3)ab	1.8 (0.7) <sup>ab</sup>	3.4 (0.9) <sup>a</sup>	F= 4.17, <i>P</i> = 0.06
Phytophagous/Saproxylic	1.3 (0.3)	0.9 (0.3)	1.2 (0.3)	1.3 (0.3)	F= 0.39, <i>P</i> = 0.8
Phytophagous/Fungivore	15.6 (3.4)	15.1 (3.9)	8.6 (2.0)	10.6 (3.0)	F= 1.86, <i>P</i> = 0.2
Predator	5.3 (1.0)	5.7 (1.2)	4.4 (0.7)	3.2 (0.7)	F= 0.85, <i>P</i> = 0.5
Sap/Sugar	9.2 (1.6)	9.2 (1.7)	11.7 (1.3)	14.6 (1.3)	F= 1.63, <i>P</i> = 0.3
Scavengers	0.4 (0.3)	0.3 (0.1)	0.6 (0.3)	0.3 (0.1)	F= 0.11, <i>P</i> = 0.9
Coprophagous**	9.3 (1.2) <sup>ab</sup>	11.6 (1.6) <sup>a</sup>	7.3 (1.7) <sup>b</sup>	9.0 (1.1) <sup>ab</sup>	F= 3.56, <i>P</i> = 0.08
Coprophagous/Carrion	1.3 (0.6)	0.8 (0.5)	1.8 (0.5)	2.3 (0.7)	F= 0.63, <i>P</i> = 0.6
Fungivore/Saproxylic	4.7 (1.7)	3.8 (1.1)	1.3 (0.3)	1.3 (0.2)	F= 0.98, <i>P</i> = 0.5
Scavenger/Saproxylic	2.2 (0.9)	2.4 (1.0)	1.9 (0.8)	1.7 (0.6)	F= 0.29, <i>P</i> = 0.8

**Table 13.** Mean ( $\pm$ SE) Shannon-Wiener diversity indices (H') and richness of genera/species (S) within four insect families captured at Green River Game Land, Polk County, NC, 2014-2016. Treatments were a high-severity burn (MB) and low-severity burn (B), each followed by three subsequent burns; a mechanical understory removal (two applications) (M); and controls (C) (n = 3 per treatment). See Table 1 for repeated treatment implementation and invertebrate sampling schedules. Genera/family with an \*\* indicate a significant difference at P  $\leq$  0.1. Within each genera/family, treatments with different superscripted letters within a row are significantly different.

						Test Statistics
	Beetle Family	MB	В	M	$\mathbf{C}$	(df = 2,3)
H'	Carabidae	0.429 (0.1)	0.400 (0.1)	0.356 (0.1)	0.410 (0.1)	F= 0.41, <i>P</i> = 0.76
	Cerambycidae	0.147 (0.07)	0.135 (0.07)	0.283 (0.09)	0.148 (0.09)	F= 1.61, <i>P</i> = 0.35
	Cleridae	0.154 (0.09)	0.031 (0.03)	0.166 (0.08)	0	F= 2.12, <i>P</i> = 0.28
	Scarabaeidae	0.782 (0.06)	0.777 (0.1)	0.622 (0.1)	0.525 (0.09)	F= 1.81, <i>P</i> = 0.32
S	Carabidae	1.5 (0.3)	1.8 (0.3)	1.4 (0.2)	1.6(0.4)	F=1.72, P=0.33
	Cerambycidae	0.8 (0.2)	0.7 (0.2)	0.9 (0.2)	0.9 (0.2)	F= 0.43, <i>P</i> = 0.75
	Cleridae	0.6 (0.2)	0.4 (0.1)	0.9 (0.2)	0.1 (0.08)	F= 4.29, <i>P</i> = 0.13
	Scarabaeidae**	$2.9 (0.2)^{ab}$	3.1 (0.4) <sup>a</sup>	2.5 (0.4) <sup>ab</sup>	2.2 (0.2) <sup>b</sup>	F= 6.80, <i>P</i> = 0.07

#### **Discussion**

Herpetofaunal, breeding bird, and invertebrate response to fuel reduction treatments was most pronounced in MB where initial high-severity burns caused substantial changes to forest structure, and repeated burns maintained a young-forest condition. A response by some invertebrate taxa, and trends of increasing breeding bird species richness was also evident after repeated burning in B, as some canopy trees died, and structural heterogeneity increased. Repeated mechanical fuel reduction treatments alone did not substantially affect any herpetofaunal, breeding bird, or invertebrate taxa, at least in the long-term.

#### Habitat/Vegetation

Cut fuels left in place for a year prior to initial prescribed burns (2003) resulted in hot fires in MB, with heavy tree mortality within three growing seasons that continued throughout two subsequent, lower-intensity burn. Heavy tree mortality in MB created an open canopy, and abundant snags that fell to pretreatment levels within a decade. Despite some reduction after each repeated, low-intensity burn, shrubs stem densities recovered rapidly and far exceed pretreatment levels in MB, as top-killed trees and shrubs resprouted and *Rubus* spp. responded to the open conditions. In contrast, the initial burn and subsequent burns in B were relatively low-intensity, resulting in delayed tree mortality at much lower levels than in MB. Tree mortality in B was concentrated in smaller trees, but burn hotspots' also killed some larger trees, eventually creating a more heterogeneous, 'perforated' canopy structure with gaps, attracting some openforest bird species (Askins 2001), albeit at much lower levels than in MB. Leaf litter depth decreased in both B and MB after each repeated burn (except in B after the second burn), but recovered within three growing seasons as leaves dropped from deciduous trees each fall. Forest structure remained relatively static in both M and C throughout the 16-year study period; post-treatment understory reductions in M were relatively small and transient.

Restoration to an open oak woodland condition was not achieved in any treatment (Waldrop et al. 2016). Despite an open canopy structure and increased sunlight in MB, herbaceous plant cover increased only modestly (peaking at 13%), and grass cover was negligible (peaking at 3% cover, with no differences among treatments) in MB. Thick shrub cover, and (or) absence of a seedbank, likely prevented proliferation of grasses and forbs (Waldrop et al. 2016).

Because both the B and MB treatments were burned four times, we could not address whether ongoing tree mortality or snag longevity in B was initiated by the initial, low-intensity burn, or perpetuated by subsequent burns. Artman et al. (2001) reported that repeated, low-intensity annual or alternate-year dormant season burns in an Ohio hardwood forest did not cause tree mortality, at least in the short-term. In contrast, our results indicate that a single high-severity dormant season burn kills large and small trees, with immediate and profound changes in forest structure. The contrast between MB and other treatments in species richness, total density, and densities of select taxa indicates that the initial high-severity burn, followed by subsequent burns, created and maintained open, young-forest conditions suitable for many vertebrate and invertebrate fauna.

## **Reptiles and Amphibians**

Capture rates of two common lizard species were greater in MB, but capture rates of other common herpetofauna, including plethodontid salamanders, were not affected by any fuel reduction treatment even after repeated applications over several years. Increased five-lined

skinks and eastern fence lizards in the MB treatment was likely due to the open forest structure created by the high-severity burns, with increased light and temperatures at ground level providing opportunity for thermoregulation, and suitable sites for egg deposition. Other studies in upland hardwood or mixed pine-hardwood forest showed that capture rates and (or) recruitment of fence lizards and (or) five-lined skinks increases following disturbances with heavy overstory removal such as clearcut or shelterwood harvests (McCleod and Gates 1998, Greenberg et al. 2016), thin-with-burn treatments (Sutton et al. 2013, 2014), open woodland restoration (Perry et al. 2009), or large canopy gaps (Greenberg 2001). Fouts et al. (2017) reported a relationship between canopy openness and abundance of northern fence lizards (S. undulatus hyacynthinus) and skinks (Plestiodon spp.), and modeled longer daily activity periods by fence lizards in burned than unburned xeric pine-oak forest. Our results showed a marginally greater capture rate of juvenile eastern fence lizards in MB, and a similar, but nonsignificant, trend for juvenile five-lined skinks. Additionally, the capture rate of juvenile eastern fence lizards increased in MB during 2014-2016, two growing-seasons after the third burn and continuing after the fourth burn. Our long-term results show that high-severity burns with heavy canopy reduction, followed by repeated low-intensity burns, increases habitat suitability for fivelined skinks and eastern fence lizards, and may promote greater juvenile recruitment.

We documented no treatment differences in capture rates of total, adult, or juvenile American toads, or total or juvenile green frogs, although capture rates varied considerably among years and variability was high within treatments and years. We captured a large proportion of juvenile American toads and green frogs from a single B (34% and 26%, respectively) and MB (35% and 11%, respectively) experimental unit in block 1 (an additional 21% of juvenile green frogs also came from a single M unit, block 3), suggesting that capture rates for these aquatic breeders were related to proximity to breeding sites. However, radiotracked individual adult American toads captured within our study area migrated an average of 994 m to a breeding pond complex on adjacent private land, from the B and MB unit in block 1 but also from a M and C unit in block 2, a similar distance away (Pitt et al. 2013). Pitt et al. (2013) did not track juvenile recruits, but their results indicated that recruits could potentially emigrate from the pond complex to all similarly distant units. Thus, an explanation for why we captured more juvenile recruits in just two or three experimental units is elusive, unless additional, undetected breeding sites occurred nearby. Shorter-term results from the same study site indicated that that distance from known water sources (large puddles, streams, and seepages) was not a significant covariate for total anurans or American toad captures (Matthews et al. 2010). However, American toad and green frog tadpoles can metamorphose in less than 10 weeks, and thus could potentially reproduce successfully in ephemeral water sources that were not identified.

Contrary to what we predicted, we documented no significant change in capture rates of plethodontid salamanders or northern red salamanders immediately after repeated burns when leaf litter was reduced, or in the longer-term after multiple burns in either burn treatment (B and MB). Earlier, short-term results from the study site (Greenberg and Waldrop 2008) immediately following initial treatments (2003–2004) also showed no change in plethodontid salamander or northern red salamander capture rate. Subsequent results following a second burn in B and MB (2006–2007) showed no difference in plethodontid salamanders in 2006 but fewer in MB than B or C in 2007 (Matthews et al. 2010). Several other studies showed that one or two low-intensity, dormant-season burns do not adversely affect terrestrial salamanders (Ford et al. 2010, O'Donnell et al. 2015, Greenberg et al. 2016, Greenberg et al. *in press*). Importantly, the long-

term results from this study indicate that, despite some variability among some treatments within individual years (Matthews et al. 2010), terrestrial salamander abundance did not decline overall, even after four repeated burns in B and MB, and further canopy cover reduction in MB.

The absence of a significant salamander response to MB was somewhat surprising because initial high-severity burns eventually reduced live tree density and associated canopy cover to levels analogous to those following shelterwood harvests, and an open canopy was maintained through repeated burning. Most studies indicate that plethodontid salamander capture rates decrease within 1–3 years after silvicultural treatments with substantial canopy removal, such as shelterwood or clearcut regeneration harvests, relative to controls or treatments involving understory manipulations where the forest canopy remains intact (Ash 1988, Harpole and Haas 1999, Homyack and Haas 2009, Perry et al. 2009, Greenberg et al. 2016). A decrease in salamander capture rate after regeneration harvests is usually attributed to increased light, and decreased leaf litter, decreased moisture, and greater temperature at the forest floor (O'Donnell 2016), which have been suggested to promote evacuation, mortality from desiccation or starvation, retreat underground (Semlitsch et al. 2009), or reduced fecundity (Homyack and Haas 2009). We were unable to address salamander fecundity because we captured few juveniles in any treatment or controls. However, the long-term monitoring and continued capture of salamanders in all treatment units indicates that adult salamanders did not evacuate or die in substantial numbers in response to repeated prescribed burning, higher fire severity, or repeated mechanical understory reduction. Hence, we suggest that terrestrial salamanders, and northern red salamanders in particular, may be resilient to low- and high-severity dormant-season burns and repeated burning in upland hardwood forests.

We observed a clear, non-significant trend of higher salamander capture rates in B that remained consistent throughout four repeated, low-intensity burns; this trend was not evident in MB. Notably, several other studies using drift fences with pitfall traps in upland hardwood forest have also reported a greater capture rate of terrestrial salamanders immediately following low-intensity dormant-season burns (Matthews et al. 2010, Greenberg et al. 2016). This could be due to higher activity levels associated with decreased litter depth or cover and associated changes in microclimate, changes in prey availability, or expanded foraging areas (Homyack et al. 2011). In contrast, other studies reported short-term decreased surface activity and abundance, and increased use of cover objects after one (O'Donnell et al. 2015, 2016) or two (Ford et al. 2010) low-intensity dormant-season prescribed burns in upland hardwood forest, based on diurnal plot searches or coverboard methods.

Changes in surface activity, or changes in the relative number of individuals above and below ground, following some fuel reduction treatments could have influenced capture rates. Presumably, under this condition, recapture rates also should increase. However, recapture rates of commonly captured species remained negligible (range 0.9% to 12% for plethodontid salamanders and five-lined skinks, respectively) across all years and treatments; additionally, our analyses included only first-captures, or new individuals. Thus, we suggest that changes in relative capture rates among treatments, within (not among) a species, reflected the relative abundance of the local population actively moving on the forest floor surface, and was not simply a metric of change in surface activity levels by the same individuals.

## **Breeding Birds**

Breeding bird density and species richness increased (24.6% to 68%, and 40.4% to 70.2%, respectively) and remained consistently higher in MB than other treatments starting three breeding seasons after initial high-severity burns with heavy tree mortality, without any apparent

additive effects of three subsequent prescribed burns. In contrast, breeding bird density was unaffected by gradual, more subtle changes to the forest structure in B, but species richness became increasingly variable beginning three breeding seasons after initial burning, and increased modestly over time, ranging 2.5% to 82% higher than pretreatment levels beginning in 2005; statistically, species richness in B did not differ from other treatments, including MB. Total bird density was unaffected by mechanical understory removal beneath an intact canopy in M; species richness varied somewhat, but changes appeared to be unrelated to treatment applications and did not differ from species richness in C. Our results suggest that both species richness and total bird density are closely, positively associated with canopy openness in upland hardwood forest; high shrub density further contributes to higher bird density, whereas greater snag density and leaf litter depth further contribute to higher bird species richness.

Increased species richness in MB was primarily due to an influx of species associated with young, open forest and edge conditions such as eastern bluebirds, indigo buntings, eastern towhees, brown thrashers (*Toxostoma rufum*), chipping sparrows (*Spizella passerina*), American goldfinches (*Carduelis tristis*), mourning doves, red-headed woodpeckers (*Melanerpes erythrocephalus*), pine warblers, and prairie warblers (*S. discolor*) starting within three breeding seasons of the initial, high-severity burn. Density of most species associated with mature, closed canopy or interior forest conditions, such as scarlet tanagers, blue-gray gnatcatchers, scarlet tanagers, and red-eyed vireos also remained high in MB throughout the 16-year study period. However, we could not assess effects on some forest interior species of conservation concern, such as wood thrush (*Hylocichla mustelina*), due to low sample size. The presence of some overstory trees provided adequate structure for canopy-associated birds, and thick cover provided by heavy sprouting of top-killed trees and shrubs offered optimal foraging opportunity for post-fledgling bird species that are otherwise associated with mature forest (Whitehead 2003; Marshall et al. 2003).

Bird density within the cavity-nesting guild was higher in MB than other treatments. However, it was unclear whether this was directly associated with higher snag densities, or with the open conditions created by heavy tree mortality, as cavity-nesting bird density remained high in MB even as snag density decreased substantially over time. Some cavity-nesting species (Eastern bluebirds and white-breasted nuthatches) were more abundant in MB than C or M. Other cavity-nesting species, including downy woodpeckers, Eastern tufted titmouse (*Baeolophus bicolor*), Carolina chickadees, and Carolina wrens (*Thyrothorus ludovicianus*) showed treatment differences, but their densities did not differ between any treatment and C, and response trends in relation to treatments were unclear. Rush et al. (2012) did not see increases in cavity-nesting species in high-severity burns with high snag densities in the southeastern southern Appalachians, suggesting that snags may not be a limiting factor.

Shrub-nester density was also higher in MB than other treatments, starting about three breeding seasons after the initial high-severity burn and remaining higher throughout three subsequent burns. Surprisingly, shrub-nester density did not significantly change over time in M or B, and trends did not closely correspond with repeated reductions in shrub density after mechanical understory removal or burns. In contrast, some studies report short-term declines in shrub-nester density after understory reduction treatments (Rodewald and Smith 1998; Greenberg et al. 2007). Densities of several shrub-nesting species, including Eastern towhees, indigo buntings, and mourning doves increased in MB within three breeding seasons of the initial high-severity burn and remained higher than in other treatments, showing no clear response to subsequent burns. Vander Yacht et al. (2016) also reported higher occupancy of

Eastern towhees, indigo buntings, and several other disturbance-dependent species in stands with ≤14 m<sup>2</sup> BA. American goldfinch were also more abundant in MB than C. In contrast, hooded warbler (Setophaga citrina) density was lower in B than C, but did not differ from M or MB. Short-term results of this study indicated that hooded warbler density decreased immediately following implementation of all three initial fuel reduction treatments (Greenberg et al. 2007). This response was not statistically significant over the longer term encompassing multiple burns, but trends nonetheless suggest short-term decreases in density after each burn, followed by recovery; we did not see decreased density after a second mechanical understory removal. Results of other studies are equivocal, showing decreased (Artman et al. 2001), no response (Aquilani et al. 2000), or increased (Rush et al. 2012) hooded warbler density after low-severity burns. Vander Yacht et al. (2016) reported reduced hooded warbler occupancy as woodland and savanna conditions (low basal area and increased grass/forb cover) were approached. In our study, shrub recovery was rapid, and especially heavy in MB, despite temporary reductions in stem density after each fuel reduction treatment. Rush et al. (2012) noted that thick shrub cover occurring within a few years of high-severity burns supported higher numbers of some groundand shrub-nesting bird species.

An overall trend of higher tree-nester density in MB was apparent although non-significant, likely due to high pre-treatment density. Many tree-nesting species showed no response to treatments, despite substantially reduced canopy cover in MB. Others, including cedar waxwings, pine warblers and eastern wood-pewees (*Contopus virens*) increased in density after the high-severity burn in MB and, to a lesser extent and over a longer period in B. This suggests that reduced canopy cover, or increased structural heterogeneity created by burning increases habitat suitability for these tree-nesting species despite reductions in tree density. Artman et al. (2001) also reported an increase in Eastern wood-pewees after burning in hardwood forest. Increased visibility and (or) density of flying insects (Campbell et al. 2007) after burning could enhance foraging opportunity, or visibility for insect salliers such as Eastern wood-pewees.

Ground-nester density did not differ between MB and B, but was lower in MB than C or M. A clear trend of short-term decreases in ground-nester density after burns and lasting for two or three breeding seasons, was evident in both burn treatments but was not statistically significant. Among ground-nesting species responding to treatments, worm-eating warblers were marginally less abundant in MB than C and density generally decreased after burns; this trend was more pronounced in MB than B and not statistically significant. Black and white warbler density did not differ among treatments, but a pattern of short-term decreased density after burning was apparent. Ovenbirds were less abundant in MB and B than M (but did not differ from C), but did not show an immediate response to each prescribed burn, indicating that factors such as canopy- and shrub cover may also influence their response (Rush et al. 2012). Other short-term studies in upland hardwood forests also indicate that ground-nesting birds including black-and-white warblers (*Mniotilta varia*) (Aquilani et al. 2000; Greenberg et al. 2007), ovenbirds (Artman et al. 2001; Aquilani et al., 2000; Klaus et al. 2010; Vander Yacht et al. 2016) and worm-eating warblers (Artman et al. 2001; Greenberg et al. 2007; Vander Yacht et al. 2016) decrease in density or occupancy after prescribed burning, or with basal area reduction and increasing canopy openness. This pattern of short-term decreases and rapid recovery of ground-nesting birds corresponds with temporarily decreased depth and cover of the leaf litter nesting substrate after burns, and recovery as leaves drop from deciduous trees each autumn.

Other studies indicate that fire severity, time since burn, or both influence patterns of bird species occurrence in hardwood forests (Klaus et al. 2010; Rose and Simons 2016; Grundel and Pavlovic 2007). Klaus et al. (2010) also reported higher species richness in medium- and highseverity burns relative to low-severity burns or unburned southern Appalachian hardwood forest. Their study, which included higher elevations than ours, also showed a positive response to highseverity burns by many of the same young, open forest-associated species, in addition to higherelevation species such as chestnut-sided warblers (Setophaga pensylvanica) and (once) goldenwinged warblers (Vermivora chrysoptera). Our results corroborated their findings that snag density, initially very high, did not appear to be a primary driver of species richness and density, which remained high even as snag density decreased over time (Klaus et al. 2010). Based on their results showing increased bird diversity 3–6 years after a high-severity burn, Klaus et al. (2010) suggested that subsequent frequent, repeated burning could lead to decreased bird diversity by inhibiting 'habitat regeneration.' We also found a delayed increase in bird density and species richness after high-severity burning, but our results suggest that repeated burning at 3-6 year intervals in high-severity burns may help to maintain high bird species richness and density by deterring forest regrowth to canopy closure, and maintaining open forest conditions. However, our study was not designed to specifically address effects of fire frequency on forest structure or breeding bird communities, and we could not assess whether more frequent burns, or more than four repeated burns, eventually affects tree and shrub sprouting.

Several studies indicate that breeding bird response is negligible or transient after single (Aquilani et al. 2000; Greenberg et al. 2007; Klaus et al. 2010; Greenberg et al. 2014) or repeated (Artman et al. 2001) low-intensity dormant season burns, or after mechanical understory reductions (Rodewald and Smith 1998) where the overstory remains intact. Our longer-term results show that tree mortality and structural heterogeneity increased modestly in B, and breeding bird species richness became more variable in B (increasing to levels no different from MB, but also no different from C or M) over time after initial and three subsequent, low-intensity burns. Again, we could not determine whether increasing tree mortality and breeding bird species richness was associated with delayed effects of the initial burn, or due to repeated burning. Nonetheless, our results suggest that low-intensity burns or repeated burns may result in modest increases in breeding bird species richness over a longer period if they create some structural heterogeneity.

#### **Pollinating Insects**

We found greater bee abundance in B and MB treatment plots compared to C, but no difference in flower visitor abundance between M and C. These differences corresponded with the modest (in B) or substantial (in MB) reductions in canopy cover and slight increases in percent cover of herbaceous plants in both burn treatments compared to M or C, where canopy cover remained intact (Waldrop et al. 2016). Other studies indicate that overstory reduction by thinning can increase richness and abundance of pollinators (Taki et al. 2010), suggesting that light levels and understory plant response may be a driving factor. Additionally, burning in B and MB plots reduced leaf litter and duff depth (Waldrop et al. 2016), leaving some exposed soil that provided suitable nesting habitat for many insects. The reduction in duff depth (Waldrop et al. 2016) likely had profound effects on abundance of ground-nesting *Lasioglossum* that usually need exposed soil for nest construction. Previous studies of forests in the southeastern US have found pollinators and flower visitors to be more abundant in fire-managed forests with a diverse understory of herbaceous and woody plants (Campbell et al. 2007, Hanula et al. 2015). Overall, bees and other flower visitors prefer forests that are open and lack dense shrub layers irrespective

of geographic region (Hanula 2016). In some cases, prescribed burning can maintain forests with reduced shrub cover and a greater diversity of herbaceous plant species compared to unmanaged forests (Campbell et al. 2007).

Our earlier study at the same study sites (Campbell et al. 2007) showed that abundance and species richness of bees and other flower visitors was greater in MB than all other treatments within one year of the initial high-severity burns (Campbell et al. 2007). Here we demonstrate that repeated burning resulted in increased in bee abundance and bee genera within B, as well. Thus, many of the longer-term effects of forest management may not be immediately apparent, and repeated burning over an extended period may be needed before changes occur.

Common wasp genera/families showed no responses to the treatments in the understory. Although *Vespula* and other wasps can sometimes pollinate some forest plants (Smith-Ramirez et al. 2005; Cheng et al. 2009), they are primarily predators. The lack of treatment effects on wasps in our study was likely driven by the availability of suitable nesting and nutrient resources within plots. The majority of wasps in our collections construct nests in the ground, although bare soil is not required by all taxa (e.g. *Vespula*). Moreover, it is likely that floral resource availability was less important to wasps than bees; while adult wasps are known to utilize nectar and pollen resources, larvae primarily feed on other invertebrates collected by the adults. Thus, prey items for wasps were likely sufficiently abundant in all fuel reduction treatments.

We did not detect any differences in the abundance of common bees, wasps, or other flower visitors among treatments within the midstory. Whereas burning in B and MB reduced canopy cover and increased herbaceous plant cover somewhat, overall tree species composition within the midstory was largely unaffected by the treatments (Waldrop et al. 2016). This suggests that midstory nectar/pollen resource availability, and prey items for wasps was similar among treatments.

Our results indicated that several genera and families exhibited clear preferences for particular vertical strata within the forest. Pompilidae nest in the ground and primarily hunt ground spiders to feed their young (Borror et al. 2005), explaining their higher abundances within the forest understory. Hesperiidae primarily utilize grasses and other herbaceous vegetation for larval food, and adults visit flowering herbaceous plants for nectar (Opler and Malikul 1998), explaining their greater abundance on the forest floor compared to the midstory.

Augochlora pura, Vespula spp. and D. maculata were more abundant in the midstory than forest floor collections. Over 90% of A. pura were collected within the midstory in this study; this observation is consistent with Ulyshen et al. (2010) who also reported that A. pura dominated bee canopy captures in a deciduous forest in Georgia. Augochlora pura typically nests in rotting logs and stumps of trees (Stockhammer 1966) and therefore, are probably not utilizing the canopy for nesting structure. Although the majority of deciduous trees in our study area are wind pollinated, some nectar/pollen abundant trees are partially insect pollinated (e.g. Acer) (Gabriel and Garrett 1984). Therefore, A. pura may be attracted to flowering midstory trees during some spring and summer months, and to sap and honeydew (Ulyshen et al. 2010), that are more abundant in forest canopies (Moran and Southwood 1982). Taken together, these data suggest that A. pura may be a canopy forager, allowing it to take advantage of a food niche that is underutilized by other bees. Its propensity for nesting in dead wood on the ground, a rare behavior among halictids, may allow this species to become especially prolific within forested habitats. Unlike Sobek et al.'s (2009) finding that the canopy of a temperate forest in Germany did not serve as important bee habitat, our results suggest that some bee species, and A. pura in particular, potentially utilize the canopy for foraging habitat.

Vespula spp., a genus of ground nesting wasps, was captured 70% more frequently in the midstory than the forest floor. Ulyshen et al. (2011) also reported that Vespula preferentially utilized the canopy compared to the forest floor of a deciduous forest in Georgia. We also found Dolichovespula maculata, a canopy/tree nesting species, to be more abundant in our midstory samples than the forest floor, likely due to a higher relative abundance of their preferred prey items within the canopy (Sobek et al. 2009). However, we cannot rule out the possibility that D. maculata were browsing the midstory for sap or honeydew.

#### **Beetles**

Beetle families responded differently to the fuel reduction treatments; this was not surprising, given the high diversity of our beetles in our collections. More important ecologically, abundance of most beetle families and guilds did not differ among the treatments. This is of particular interest because many of these serve as the prey base for multiple wildlife species, and contribute to nutrient cycling in forested habitats. Reduced canopy cover, increased light, and possibly drier forest floor conditions could have resulted in less fungal growth and fewer fungivorous beetles in MB. In contrast, phytophagous beetles tended to be more abundant in MB and B compared to C or M, likely due to higher herbaceous plant cover resulting from prescribed burning and higher light levels (Waldrop et al. 2016). Mordellidae, which feed on nectar and pollen as adults, were more abundant in MB and B than C or M. This response was likely due to increased cover, and most likely flowering of herbaceous vegetation after repeated burning and reductions in canopy cover. In contrast, Nitidulidae, many of which feed on fungi, were more abundant in C than B or MB, potentially due to the cooler, shadier, and moister conditions at the forest floor that promoted fungal growth relative to the open, drier conditions in burned sites.

The relative abundance of most beetle families and guilds were not affected by the fuel reduction treatments. Our results are generally consistent with earlier studies on the overall arthropod community (Greenberg et al. 2010), and on saproxylic beetles (Campbell et al. 2007) conducted within the same study sites after initial treatment application. However, Greenberg et al. (2010) found no treatment response by any of the 11 Coleoptera families examined, including three families (Scarabaeidae, Staphylinidae, and Nitidulidae) that did exhibit a treatment response in our study. The abundance trends we found with Scarabaeidae, Staphylinidae, and Nitidulidae could have been due to the multiple rounds of treatments applied over the study period. Indeed, other groups of arthropods require multiple forest disturbances or management treatments before effects on abundance or richness are detected (Campbell et al. *in press*). Apigian et al. (2006) also examined beetle assemblages within forest stands that were thinned, burned, or had combinations of thinning and burning in a mixed conifer forest in the Sierra Nevada mountains of California. They also reported that some groups of Coleoptera differed in abundance among those treatments. However, these responses did not show any general pattern, and most significant changes were taxon-specific (Apigian et al. 2006).

Although our trapping methods captured numerous beetles (> 7000), our methods could have been biased against certain taxa. Although colored pan traps have been used to capture Coleoptera (Campbell et al. 2007) they are not a standard collecting method. Despite our unconventional methodologies, any trap biases would have been equal among the treatments. We examined some beetles at a species or genus level, but most analyses were performed at the family level, potentially masking treatment effects on some genera or species. Additionally, we collected only one or two individuals of several genera or species within our focal families that

we could not analyze statistically due to low sample. In spite of these potential biases or masking effects, several beetle families and feeding guilds showed preferences for one or more fuel reduction treatments or controls.

# **Management Implications**

Our results highlight the importance of creating and maintaining structural diversity within and among forest stands to create suitable conditions for vertebrate and invertebrate species with varying habitat requirements. Management using prescribed burning can create and maintain complex forest structure. Repeated, low-intensity burns can develop a 'perforated' forest canopy over time by killing a few trees, and by temporarily reducing shrub density and leaf litter depth. High-severity burns that kill many trees can create an open, young-forest structure with abundant snags; leaf litter and shrub density are temporarily reduced, but shrub density may eventually increase as top-killed trees and shrubs resprout, and *Rubus* proliferates. This young forest condition can be maintained through repeated lower-intensity burns. Our results indicate that many common breeding bird, amphibian, and invertebrate species tolerate a wide range of forest conditions. Others, including some lizard species, ground-nesting bees and other flower-visiting insects, some Coleoptera taxa, and many breeding bird species may benefit from the open canopy, dense shrub cover (within a few years after burning), or reduced leaf litter created by burns. Other species, such as some ground-nesting bird and invertebrate taxa may temporarily decline after burning, emphasizing the importance of leaving areas of mature, unmanaged forest within a managed forest matrix. Our study illustrates the importance of longterm studies that can address potentially delayed responses to forest management, and potential additive effects of repeated management activities. Different responses among species emphasize the importance of including multiple taxa when assessing effects of forest management on wildlife, and give perspective on how the definition of forest health may vary depending on target taxa (Harper et al. 2016).

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# **Appendix A: Contact Information for Key Project Personnel**

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# **Appendix B: List of Completed or Planned Scientific or Technical Publications and Science Delivery Products (since 2014)**

### Tours (14 total):

10/30/2014 University of Tennessee Wildlife class 10/28/2015University of Tennessee Wildlife class 10/26/2016 University of Tennessee Wildlife class 10/27/2017 University of Tennessee Wildlife class 9/7/2017 North Carolina Prescribed Fire Council 8/31/2016 USDA Forest Service 10/23/2016 North Carolina Chapter of the Fire Learning Network - Also 4 additional tours, names of groups not recorded

## **Articles in peer-reviewed journals:**

#### **Published**

Campbell J.W., C.C. Vigueira, P.A. Vigueira, J.E. Hartgerink, and C.H. Greenberg. 2017. The use of root plates for nesting sites by *Anthophora abrupta* may be common within forested habitats. Florida Entomologist 100:488-490.

Campbell, J.W., P.A. Vigueira, C.C. Vigueira, and C.H. Greenberg. *In press*. The effects of repeated prescribed fire/thinning on bees, wasps, and other flower visitors in the understory and midstory of a temperate forest in North Carolina. Forest Science.

Campbell, J.W., O. Keller, C.C. Vigueira, P.A. Vigueira, E. Waite, and C.H. Greenberg. *In review*. The Response of Coleoptera to Repeated Application of Prescribed Fire and Other Fuel Reduction Techniques in the southern Appalachian Mountains. Canadian Journal of Forest Research.

Greenberg, Cathryn H.; Waldrop, Thomas A.; Tomcho, Joe; Phillips, Ross J.; Simon, Dean. 2014. Bird response to fire severity and repeated burning in an upland hardwood forest. pp. 81-84, In: Waldrop, Thomas A., ed. Proceedings: Wildland fire in the Appalachians: discussions among fire managers and scientists. Gen. Tech. Rep. SRS-199. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 206 p.

Greenberg, C.H., C.E. Moorman, C.E. Matthews-Snoberger, T.A. Waldrop, D. Simon, A. Heh, and D. Hagan. 2017. Long-Term Herpetofaunal Response to Repeated Fuel Reduction Treatments. Journal of Wildlife Management. Early View (DOI) - 10.1002/jwmg.21402

Waldrop, T.A., Hagan, D.L., and Simon, D.M. 2016. Repeated application of fuel reduction treatments in the southern Appalachian mountains, USA: implications for achieving management goals. Fire Ecology 12:28-47.

# In preparation

Campbell, J.W., M. Milne, P.A. Vigueira, C.C. Vigueira, and C.H. Greenberg. *In preparation*. The response of ground-dwelling arthropods to repeated application of prescribed fire and other fuel reduction techniques in the southern Appalachian mountains of North Carolina.

Greenberg, C.H., J. Tomcho, A. Tomcho, J.D. Lanham, T.A. Waldrop, D. Simon, and D. Hagan. *In Preparation*. Long-term Avian Response to Fire Severity, Repeated Burning, and Mechanical Fuel Reduction in Upland Hardwood Forest.

#### **Scientific Conference Presentations and Published Abstracts**

Campbell, J.W., P.A. Vigueira, C.C. Vigueira, and C.H. Greenberg. The effects of repeated prescribed fire and thinning on bees, wasps, and other flower visitors in the understory and midstory of a temperate forest in North Carolina. Special Symposium, Fire Role in Restoration of Wildlife, Ecosystem Resilience and Ecosystem Services. 7th International Fire Ecology and Management Congress, 11-30, 2017, Orlando, Florida. Oral presentation and published abstract.

Greenberg, C.H., C.E. Moorman, C.E. Matthews-Snoberger, T.A. Waldrop, D. Simon, A. Heh, and D. Hagan. Herpetofaunal response to fire severity, repeated burning, and mechanical fuel reduction in hardwood forest. Special Symposium, Fire Role in Restoration of Wildlife, Ecosystem Resilience and Ecosystem Services. 7th International Fire Ecology and Management Congress, 11-30, 2017, Orlando, Florida. Oral presentation and published abstract.

Greenberg, C.H., J. Tomcho, A. Tomcho, J.D. Lanham, T.A. Waldrop, D. Simon, and D. Hagan. Breeding bird response to fire severity, repeated burning, and mechanical fuel reduction in hardwood forest. Special Symposium, Fire Role in Restoration of Wildlife, Ecosystem Resilience and Ecosystem Services. 7th International Fire Ecology and Management Congress, 11-30, 2017, Orlando, Florida. Oral presentation and published abstract.

Vigueira, P.A., J.W. Campbell, C.C. Vigueira, and C.H. Greenberg. 2016. Effects of continued prescribed fire and fire surrogates on insect pollinators of the Blue Ridge Province in North Carolina. International Congress of Entomology, 2016. Orlando, FL. *Poster and published abstract*.

#### **Technical Workshops for Natural Resource Professionals**

Greenberg, C.H. 2015. Prescribed Fire and Wildlife. National Advanced Silviculture Program (NASP) – Upland Hardwoods Silviculture Module. 13 May, 2015, Bent Creek Experimental Forest.

Greenberg, C.H. 2017. Fire and Wildlife: A Historical Context. National Advanced Silviculture Program (NASP) – Upland Hardwoods Silviculture Module. 22 May, 2017, Bent Creek Experimental Forest.

### **Presentations to Lay Audiences**

Greenberg, C.H. 2017. Fire and Wildlife: A Historical Context. University of North Carolina Asheville, GreenFest. Oral presentation and panel on 2016 wildfires in western North Carolina. 3-28-2017.

Greenberg, C.H. 2017. Fire and Wildlife: A Historical Context. MountainTrue and Asheville Citizen-Times sponsored, Highlands Brewery. Before We Burn Again: A Panel on the Future of Wildfires in WNC. Oral presentation and panel. 4-3-2017.

Greenberg, C.H. 2017. Fire and Wildlife: A Historical Context. Elisha Mitchell Chapter of the Audubon Society. Oral presentation 11-21-2017.

# **Appendix C: Metadata**

The data collected during this project (2014-2016) include vegetation, fuels (collected in 2014 and 2015) herpetofaunal (2014-2016), breeding bird (2014-2016), and invertebrate data (2014-2016) collected in three replicate units each of four fuel reduction treatments (control, mechanical, burn, mechanical + burn), for a total of 12 experimental units. Bird data includes all birds detected within three 50 m radius point counts per experimental unit, each visited three times each breeding season. Herpetofaunal data include all reptile and amphibian individuals captured during spring-summer trapping periods in the same number of drift fence - pitfall trap funnel trap arrays per experimental unit (2 or 3 arrays, depending on the year; array-nights also included in data files). Pollinating insect data include number of individuals per taxa captured in colored pan trap sets. A set consisted of one red, blue, white, and yellow bowl placed at each corner of a 66 cm square of metal remesh, placed on the forest floor and in the midstory. A set (forest floor and midstory) was placed at each of 2 locations > 50 m apart within each experimental unit. Beetles, ants (not used in this report), and other ground-dwelling invertebrates were sampled using two pitfall trap arrays spaced > 50 m apart within each treatment unit. Arrays consisted of a 118 ml cup filled half-way with soapy water, with three 30 cm long aluminum flashing drift fences trenched into the ground, radiating from the center of the cup and oriented at 120° to the neighboring drift fence. Bowl sets and pitfall traps were deployed for 72 hours at monthly intervals (May/June- September/October) from 2014-2016, for a total of 11 colored pan trapping periods (three in 2014; four in 2015 and 2016) and 12 pitfall trapping periods (four each year) during the three year study. Vegetation was measured in 2014, 2015, and 2016. Overstory trees and snags (>10 cm dbh), and shrubs/saplings (trees >1.4m tall and < 10 cm dbh; not used in this report) were measured within 10, 0.05-ha (10 x 50 m) plots located at 50 x 50 m intervals starting from a randomly selected grid-point origin within each experimental unit. Shrub stem density was measured within 20, 1m<sup>2</sup> quadrats within each vegetation plot. Down dead woody fuels were measured only in 2014 and 2015 using the planar intercept method (Brown 1974). Three, 15.2 m transects were established approximately 2 m from each grid-point in a randomly selected direction. Leaf litter depth was measured using a meter stick at 3.0, 7.6, and 12.2 m from the origin of fuel transects. All files are in EXCEL and include a metadata worksheet that describes each data field. The data and accompanying metadata will be archived in the JFSP-recommended repository, the Forest Service Research Data Archive, upon publication of remaining peer-reviewed papers presenting the data. Files are:

Birds – 2014 2015 2016
Fuels Transects 2014
Fuels Transects 2015
Herpetofauna – 2014 2015 2016
Invertebrates – Ants 2014 2015 2016
Invertebrates – Beetles 2014 2015 2016
Invertebrates – Pollinators 2014 2015 2016
Vegetation Overstory 2014
Vegetation Overstory 2015
Vegetation Overstory 2016

Vegetation 1m2 (understory) 2014 Vegetation 1m2 (understory) 2015 Vegetation 1m2 (understory) 2016 Vegetation ShrubSaplings 2014 Vegetation ShrubSaplings 2015 Vegetation ShrubSaplings 2016